PREDICTING NUTRIENT AND DISSOLVED OXYGEN CONCENTRATIONS IN THE SPOKANE RIVER UNDER FUTURE DEVELOPMENT AND CLIMATE CONDITIONS

by

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ABSTRACT

Waste load allocations in the United States have traditionally been based on historical data. Total Maximum Daily Load (TMDL) policies and stormwater permits are often based on steady-state low flow analysis without considering unsteady flow and pollutant conditions prevalent during stormwater events. Questions surrounding remedial investments for conditions that do not exist during low flow events have been raised. Furthermore, the frequency of violations under various pollutant removal scenarios has been hard to determine.

The Spokane River TMDL used a 2001 low flow event as the basis for planning multimillion dollar investments in wastewater treatment systems and stormwater plan. This study modified the original CE-QUAL-W2 model to simulate hydrology and water quality over a 1999–2009 period with attention to phosphorus, nitrogen, and dissolved oxygen. Calibrating and applying the model for an extended period enabled better prediction of nutrient dynamics under varying flow conditions, which simplified the investigation of permissible nutrient levels. Model results showed that hydrologic conditions outside the low flow period may be cause for concern. Violation of water quality standards occurred outside the 2001 duration with both phosphorus and nitrate concentrations being much higher in years with higher flows. Results demonstrated the need for better understanding of the influence of nonpoint sources on Spokane River-Long Lake water quality, and that

additional information concerning stormwater inputs and nutrient cycling would permit better decisions in the future.

Finally, the calibrated CE-QUAL-W2 model was applied to the Spokane River-Long Lake as a case study to simulate water quality changes in response to various climate change and population growth scenarios. Model results indicated the disproportionate increase of streamflows during winter, which can overwhelm the existing stormwater flow and pollution control infrastructure through exceedingly high nutrient loadings. Moreover, model simulations revealed extreme nature of climate change impacts, where streamflow increase seemed to have some positive effect on water quality at surface layers, but nutrient and dissolved oxygen concentrations in the deeper layers did not experience any improvement. While the current TMDL proposes a 50% reduction of nonpoint loading, results indicated that this will not be adequate. Such multifaceted nature of climate change effect on water quality makes management decisions more complex for water managers, and indicates the need for revisiting the citations of point and nonpoint source loading reduction in the existing TMDL based on low flow analysis, targeted towards meeting dissolved oxygen standards in the Spokane River-Long Lake system.

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CHAPTER 1

INTRODUCTION

1.1 Problem Description

Water of sufficient quantity and quality is essential for drinking, agriculture, energy production, navigation, recreation, ecosystem functions, and manufacturing. As water resources deficiencies due to population increases and climate change effects become even more prevalent, societies around the world are beginning to understand that sustainable economic growth can be extremely sensitive to variations in the storage, fluxes, and quality of water at the land surface (Lettenmaier et al., 2008). In response to current quantity and quality concerns in the U.S., states have increased protective strategies such as adoption of green infrastructure practices (USEPA, 2010; Foster et al., 2011) and development of Total Maximum Daily Loads (TMDL) (Berger et al., 2002; Berger et al., 2003; Annear et al., 2005; Moore and Ross, 2010).

Numerous researchers have concluded that climate change is likely to increase water demand while shrinking water supplies (Groisman et al., 2001; Hodgkins et al., 2005; Novotny and Stefan, 2007; Tu, 2008; Tu, 2009). Higher air temperatures, earlier snowmelt, and potential decreases in summer precipitation will likely increase the risk of dry years (Field et al., 2007). Furthermore, many areas of the world are currently facing water supply issues, and demands will continue to rise as populations grow. This shifting balance will challenge water managers to simultaneously meet the needs of growing communities, sensitive ecosystems, farmers, ranchers, energy producers, and manufacturers (Kundzewicz et al., 2007; Hatfield et al., 2008).

While numerous studies have demonstrated deleterious impacts on water quantity associated with climate change and population growth (Field et al., 2007; Bates et al., 2008; Katrina et al., 2012), these two factors also can have harmful impacts on surface water quality (Andersen et al., 2006; Kaste et al., 2006; Whitehead et al., 2009; Green et al., 2011; Stuart et al., 2011; Jin and Stridhar, 2012). Changes in the quantity, timing, intensity, and duration of precipitation and runoff (Backlund et al., 2008; Hatfield et al., 2008) coupled with higher water temperatures (Ficke et al., 2007; Kundzewicz et al., 2007; Nõges et al., 2010; Klose et al., 2012; Andersen et al., 2006; Solheim et al., 2010) can adversely affect water quality including increasing contaminant concentrations (Kaste et al., 2006; Bates et al., 2008; Whitehead et al., 2009; Katrina et al., 2012), enhancing the potential for toxic algal blooms (Whitehead et al., 2009; Solheim et al., 2010; Verweij et al., 2010; Paerl and Paul, 2012), and reducing dissolved oxygen levels (Fu et al., 2007a; Fu et al., 2007b; Backlund et al., 2008; Hatfield et al., 2008; Karl et al., 2009), eventually decreasing the self-purification capabilities of rivers (Ficke et al., 2007). The severity of these problems is more evident when the seasonal cycles are looked at (Bouraoui et al., 2002; Chang 2004; Tsvetkova, 2013).

Increased pollutant concentrations and lower dissolved oxygen levels will result in waterbodies not meeting water quality standards and, therefore, being listed as impaired waters (USEPA, 2008). This suite of water quality effects will increase the cost of meeting water quality goals for both consumptive and environmental purposes (Gleick et al., 2000), as sewage treatment alone may not be sufficient to maintain low nutrient levels under climate change impacts (Hadjikakou et al., 2011). Discharge permits and nonpoint pollution control programs may need to be adjusted to reflect the changing conditions (Moore and Ross, 2010). Moreover, mitigation solutions may not accurately reflect future conditions (IPCC, 2012), and thus money may not be targeting the most effective remediation strategy. Apart from climate change, population increases and land use changes will result in more stormwater runoff and discharges from wastewater treatment facilities, which will have added effects on the surface water quality (Miserendino et al., 2011; Tu, 2011; Yu et al., 2013).

Significant gaps still exist in the knowledge about the combined impacts of climate change and population growth on water quality (Witte et al., 2012; Crossman et al., 2013; Andersson et al., 2015). Water resources management decisions are being made obligating billions of dollars of expenditures without fully understanding the implications of climate, land use, and population changes on future pollutant impacts (Grafton et al., 2013). Traditionally, low flow or steady-state analyses have been used to form management plans assuming it to be the worst case scenario (Macintosh et al., 2011). However, when nonpoint sources from stormwater runoff and nutrient cycling represent significant loading conditions, decisions based on low flow analysis may provide inadequate or incomplete information and thus invalidate mitigation decisions (Yaeger et al., 2014). There is a need to improve the understanding and modeling of climate and population changes with respect to water quality at scales that are relevant to decision-making in water management, particularly for large-scale water management infrastructure, the design of which is dependent on assumptions based on climate stationarity (Stakhiv and Stewart, 2010).

1.2 Goals and Objectives

The overarching goal of this project is to improve our understanding of how changes in watershed conditions due to population growth and climate change may alter management decisions with respect to both point sources (called the wasteload allocation) and nonpoint sources (called the load allocation) in the TMDL process. To help accomplish this goal, these specific objectives will be completed:

1. Expand and calibrate an existing low flow TMDL model to encompass a 10-year hydrologic period (1999-2009) that includes the impacts of stormwater runoff;
2. Evaluate how these long-term simulations compare to the low flow analysis in terms of nutrient reduction requirements;
3. Examine the impacts of population projections on point loads under climate induced flow conditions;
4. Investigate how climate change impacts management decisions related to point and nonpoint source nutrient loading reduction.

These objectives were achieved by developing a long-term hydrodynamic water quality model of the Spokane River watershed encompassing a variety of hydrologic conditions. The model is an extension of an existing CE-QUAL-W2 model developed for predicting phosphorus reductions necessary to control excessive algae blooms and low dissolved oxygen (DO) levels during a low flow year (Slominski et al., 2002; Berger et al., 2003; Moore and Ross, 2010). CE-QUAL-W2 is a two-dimensional, laterally averaged, hydrodynamic water quality model developed by the US Army Corps of Engineers capable of handling the major chemical and biological processes like effects of dissolved oxygen on atmospheric exchange, photosynthesis, respiration, organic matter decomposition, nitrification, chemical oxidation of reduced substances; uptake, excretion, and regeneration of phosphorus and nitrogen and nitrification-denitrification under aerobic and anaerobic conditions; carbon cycling and alkalinity-pH-CO2 interactions; trophic relationships for total phytoplankton; accumulation and decomposition of detritus and organic sediment; and coliform bacteria mortality (Annear et al., 2001).

The CE-QUAL-W2 Spokane River model was developed as a case study to simulate and examine the water quality changes in response to various climate and population growth change scenarios, with particular attention to phosphorus, nitrogen, dissolved oxygen and river temperature. The model was calibrated for 1999-2009 with the available data. The calibrated model was used to simulate Spokane River water quality during 2040-2050 considering climate change and population growth scenarios. Projected streamflows were provided by University of Washington Climate Impacts Group.

The unique aspect of this study is that it uses long term water quality modeling to encompass varying hydrologic conditions which will produce a better understanding of the relative importance of the river's point and nonpoint sources in nutrient cycling under the existing and future climate and population growth scenarios. It will demonstrate the need to encompass a broader range of flow and water quality concerns into future ecosystem evaluations. Results of the case study developed as a demonstration of the impacts will be valuable for water managers to redefine any necessary modifications of the Spokane River wasteload and load allocations.

This study is divided in five chapters starting with Chapter 1 which provides the rationale for this work and the objectives. Chapter 2 describes the study area, and contains a literature review on the water quality management efforts in the Spokane River; model selection, background, and its existing application. Chapter 3 describes in detail the data collection and model development for observed and projected time periods. Details on model calibration, and results from climate change scenario simulations and alternative scenarios are presented in Chapter 4. Finally, Chapter 5 provides a discussion on the results and conclusions of this study.

CHAPTER 2

CLIMATE CHANGE IMPACTS ON WATER QUALITY, STUDY AREA, AND MODEL BACKGROUND

2.1 Impacts of Climate Change on Water Quality

Climate change is expected to have a wide range of impacts on surface water quality. Water quality can suffer due to the changes in quantity, timing, and intensity of streamflow (Fu et al., 2007a; Fu et al., 2007b; Karl et al., 2009), rises in water temperature (Ficke et al., 2007; Kundzewicz et al., 2007; Nõges et al., 2010; Klose et al., 2012; Andersen et al., 2006; Solheim et al., 2010), and excessive algal bloom (Whitehead et al., 2009; Solheim et al., 2010; Verweij et al., 2010; Paerl and Paul, 2012). Furthermore, these impacts are anticipated to vary considerably temporally and spatially indicating that site specific solutions may be required. For example, future projections of less snowpack and earlier snowmelt in the northwestern part of the United States will likely cause less water availability during the summer months, when demand is highest (CCSP, 2008) and when water quality concerns are generally the greatest.

Increases in runoff volumes can add more sediments, nutrients, pollutants, animal waste, and other materials into water bodies making them impaired or worsening an existing problem (CCSP, 2008; California Water Plan, 2009). For instance, additional climate-induced runoff can significantly increase nutrient concentrations and fluxes (Kaste et al., 2006) that can stimulate growth of benthic algae and macrophytes in river systems (Staehr and Sand-Jensen, 2006; Millican et al., 2008; Moore and Ross, 2010; Katrina et al., 2012). Neff et al. (2000), Bouraoui et al. (2002), Hatano et al. (2005), and Andersen et al. (2006) each concluded that projected increases in streamflows due to climate change impact will result in increases in total nitrogen and total phosphorus in surface waters.

Seasonal analysis of projected surface water nutrient concentrations predicts higher nutrient levels in winter due to the higher flow predictions in winter and spring (Chang et al. 2001; Bouraoui et al. 2002, Chang, 2004). Furthermore, increased occurrence of low flows in summer can cause nutrients and contaminants to become more concentrated due to decreased contaminant dilution capacity (Bates et al., 2008; Whitehead et al., 2009; Katrina et al., 2012). Kaste et al. (2006) predicted increase in nitrate and phosphate concentration during summer low-flow conditions due to decrease in stream's dilution capacity and higher residence time. Contrary to these studies, Arheimer et al. (2005) and Chang et al. (2015) found that scenarios with the highest water discharge were connected to lower increase in nutrient concentration levels, probably due to the dilution effect in the river system.

Rising water temperatures due to climate change will impact surface water quality by increasing primary production, organic matter decomposition, and nutrient cycling rates in the streams (Ficke et al., 2007; Kundzewicz et al., 2007; Nõges et al., 2010; Klose et al., 2012). Whitehead et al. (2009) reviewed climate impacts on surface water quality and found that lower flows and reduced velocities, coupled with increasing water temperature, will enhance the potential for toxic algal blooms and reduce dissolved oxygen levels.

Carmichael et al. (1996), Solheim et al. (2010), and Golosov et al. (2012) also predicts substantial reductions in dissolved oxygen concentrations due to higher temperatures. Increase in water temperature is also associated with increase in summer phosphorus concentrations (Field et al., 2007). Increased nitrification and BOD oxidation rates in summer (Cox and Whitehead, 2009; Verweij et al., 2010) due to climate change may exacerbate dissolved oxygen levels as well (Cox and Whitehead, 2009; Solheim et al., 2010; Arvola et al., 2010). Cyanobacteria problems during low flows (increased residence time) are expected to become severe with climate change (Bowes et al., 2008; Verweij et al., 2010; Solheim et al., 2010; Moore and Ross, 2010; Paerl and Paul, 2012).

In addition to climate change impacts, population growth will have added effects on conductivity, nutrients, biodiversity (Miserendino et al., 2011; Tu, 2011), water temperature, pH and dissolved oxygen (Yu et al., 2013) in the surface waters. Deterioration of surface water quality can occur when discharges at existing wastewater treatment infrastructure are increased or when stormwater is discharged directly into surface water.

Considering these changing scenarios, concerns are now being raised on the management policies of existing wastewater treatment facilities. Not only can the water infrastructure be overwhelmed by the increased volumes of wastewater yield under the impacts of changing climate and population demand (Karl et al., 2009), but also the possible inadequacy of the sewage treatment facility alone to maintain low nutrient concentrations in the rivers exists (Hadjikakou et al., 2011). Population growth will result in the need for additional investment in sewage treatment works (Fisher and Stewart, 2006). Hadjikakou et al. (2011), while evaluating different policy options for maintaining river nitrate levels, revealed that sewage treatment alone may not be sufficient to maintain target nitrate concentrations under climate change impacts.

2.2 Study Area: Spokane River

The Spokane River stretches from Coeur d'Alene, Idaho to its confluence with the Columbia River in north-central Washington, draining over 6,640 square miles of land in the two states (lower 2,295 square miles in Washington State and the remainder in Idaho). The 111 mile long river flows west from Lake Coeur d'Alene across the Washington/Idaho Stateline to Franklin D. Roosevelt Lake on the Columbia River (Cusimano, 2004). The study area for this study extends 39.2 miles from the Stateline Bridge at river mile (RM) 96.0 to Lake Spokane (Long Lake) Dam at RM 33.9. Hydroelectric dams located in this reach are Upriver Dam (RM 79.9), Upper Falls Dam (RM 76.0), Monroe Street Dam (RM 73.4), Nine Mile Dam (RM 57.6), and Lake Spokane Dam (RM 33.9). The hydrodynamics of the river is also influenced by the Post Falls Dam (RM 100.8) in Idaho (Cusimano, 2004). Except for Long Lake dam, which creates the Lake Spokane (Long Lake), the rest of the dams in Washington are run-of-the river type.

Continental climate exists in the Spokane Subbasin, which is influenced by maritime air masses from the Pacific Coast. The average annual temperature is 9.4°C, with July being the warmest (21.6°C) and January being the coldest (-1.5°C) months. The area receives an annual precipitation of about 45 cm, and snowfall of about 27 cm (Northwest Power and Conservation Council, 2004). Land use in the area is heavily impacted from agricultural activities and increasing development throughout the subbasin (Northwest Power and Conservation Council, 2004). Over 35 species of fishes are found in the Spokane Subbasin. The river water is used for hydropower generation, irrigation, and sport fishing (Northwest Power and Conservation Council, 2004). Figure 2.1 shows the study area with the tributaries and point source dischargers.

Four significant events in the basin have affected Spokane River streamflows during the past. These include (1) the completion of Post Falls Dam in 1906; (2) the operation of the Spokane Valley Farms Canal, which began in 1924 and diverted water from the Spokane River upstream of the Post Falls gaging station; (3) the change in operating practices of Post Falls Dam in 1941 to raise levels of Coeur d'Alene Lake in

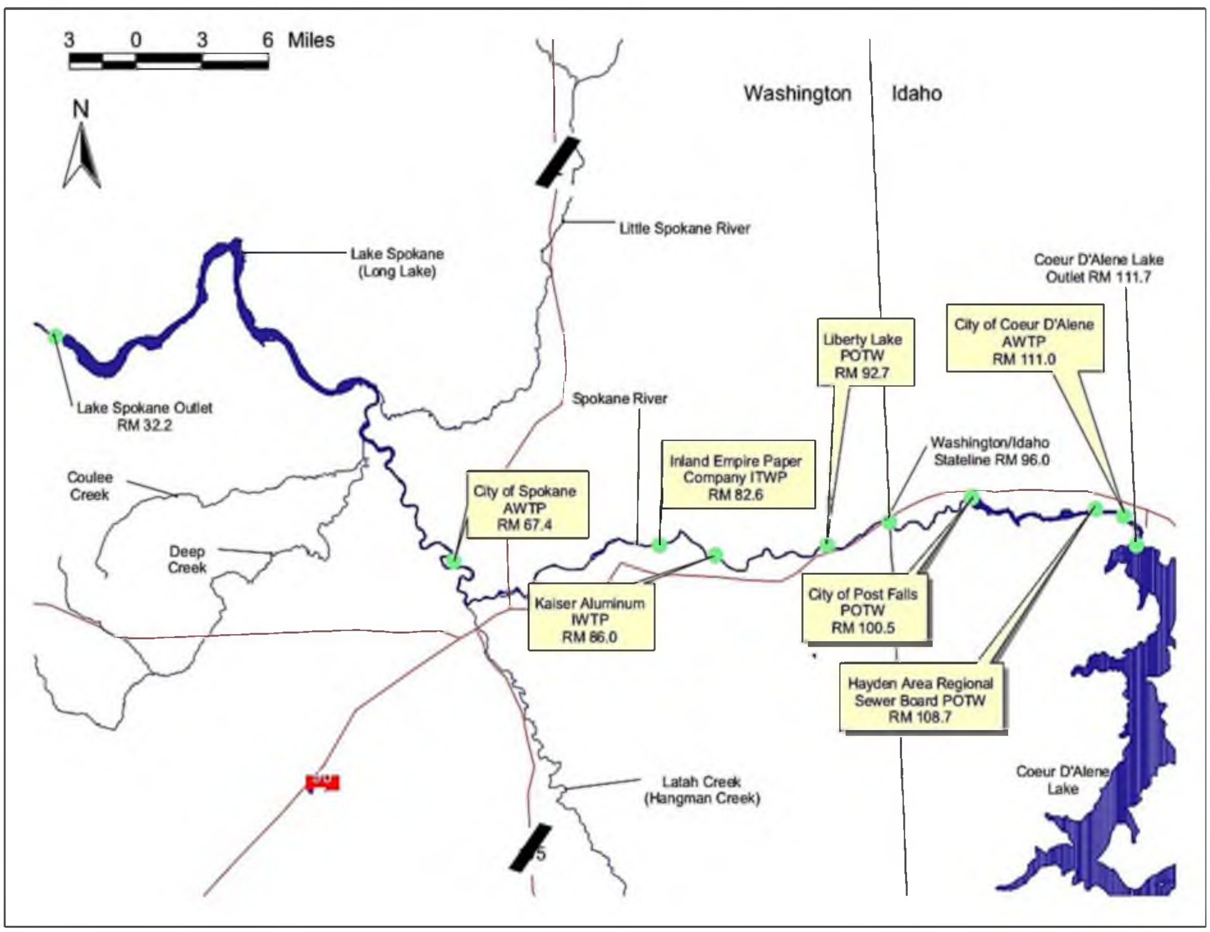


Figure 2.1 Spokane River Study Area

summer; and (4) the discontinuation of irrigation withdrawals through the Spokane Valley Farms Canal in 1967 (Hortness and Covert, 2005).

The Spokane River near Post Falls (Idaho) gage (USGS gage 12419000, RM 100.7) records a mean annual flow of 6,205 cfs in the Spokane River during 1913-2014. Near Otis Orchard (Harvard Rd.), Washington (USGS gage 12419500, RM 93.9), the mean annual flow declines to 6,083 cfs. There is a net gain in river flow from the Harvard Road gage to the City of Spokane at Monroe Street due to groundwater inflow, and the mean annual flow (1892-2014) at Spokane River at Spokane gage (USGS gage 12422500, RM 72.9) becomes 6,695 cfs. Hangman (Latah) Creek, the first major tributary, joins the Spokane River at RM 72.4 adding 231 cfs (USGS gage 12424000, 1948-2014 average) to the river. Little Spokane River joins the Spokane River at RM 56.4, and adds 302 cfs annually (1930-2014 average, USGS gage 12431000) to the river. Coulee Creek joins the Spokane River at RM 58.8, but adds very little flow to the main stream.

The highest flows in the Spokane River, resulting from snowmelt, occur from April through the beginning of June; while August and early September typically receive the lowest flows. AVISTA increases the discharge from the Post Falls Dam during the second week of September to lower the water level in Lake Coeur d'Alene (Cusimano, 2004). Gearhart and Buchanan (2000) and Cusimano (2004) described in detail the aquifer interactions of the Spokane River for reaches between the Stateline and Green Street.

Previous studies (Hortness and Covert, 2005) detected statistically significant decreasing trends in monthly mean streamflow at Spokane River near Post Falls gage (RM 100.7) for August and September. Trend analysis of annual 7-day low streamflows at this gage for the postcanal period 1968–2002 revealed a substantial decrease in low streamflows (Hortness and Covert, 2005). Statistically significant decreasing trends in monthly mean streamflows were also noticed for September at the Spokane River at Spokane gage (RM 72.9). Records at the Little Spokane River at Dartford gage showed statistically significant decreasing trends in monthly mean streamflow for September and October (Hortness and Covert, 2005). Recent and projected urban, suburban, industrial, and commercial growth, coupled with climate change implications, has raised concerns about future water availability in Spokane River, particularly during the high demand period.

There are different facilities along the river that have National Pollutant Discharge Elimination System (NPDES) permits for discharging biochemical oxygen demand and/or ammonia to the Spokane River. Nutrient loading is a concern as it indirectly impacts DO levels through increased primary productivity and consequent plant respiration and decay processes (Cusimano, 2004). The dischargers in Washington are Liberty Lake POTW (Permit No: WA-004514-4, RM 92.3), Kaiser Aluminum Industrial Wastewater Treatment Plant (IWTP) at Trentwood (Permit No: WA-000089-2, RM 86.0), Inland Empire Paper Company IWTP (Permit No: WA-000082-5, RM82.6), and the City of Spokane AWTP (Permit No: WA-002447-3, RM 67.4). Dischargers in the Idaho portion of the Spokane River also have impact on the river water quality. Each of these wastewater and industrial facilities have average monthly and weekly BOD_5 and ammonia discharge limits, the details of which can be found from Cusimano (2004). Point source discharges have been identified as the main sources of phosphorus loading to the river during the summer growing season (Patmont et al., 1985; Patmont et al., 1987; Moore and Ross, 2010). In this study, only dischargers in Washington have been incorporated in the modeling efforts.

Spokane River dissolved oxygen levels and nutrient concentrations are also affected by discharges from Hangman Creek carrying loads from communities of Cheney, Spangle, Rockford, Tekoa, and Fairfield; and Little Spokane River carrying loads from Kaiser-Mead IWTP (currently not in operation), Washington Department of Fish and Wildlife Spokane Fish Hatchery, and Colbert Landfill Superfund Site groundwater pump and treatment system. Discharge from the Hangman Creek drops rapidly to less than 20 cfs on average during July-October, contributing insignificant nutrient and organic matter loads. However, loading from Little Spokane River during this period is high relative to Hangman Creek because of significant groundwater contributions (average inflow about 400 cfs). Coulee/Deep Creeks also add loads from the City of Medical Lake (Cusimano, 2004). Knight (1998) concluded that at the current flows, the discharge from Coulee Creek will not affect the Spokane River. However, he foresaw a new growing-season phosphorus load to the Spokane River with the expansion of the system. The interaction of the river with the underlying aquifer affects dissolved oxygen and nutrient levels in the river. Nutrient loadings from point source discharges, tributaries, and groundwater interactions have all been well incorporated into the modeling effort in this study to represent the nutrient cycling in the Spokane River and Long Lake. Stormwater and combined sewer overflow discharges, although controlled, still remain a major source of nutrient and organic loading to the river.

According to the water quality criteria, the average euphotic zone concentration of total phosphorus in Long Lake reservoir shall not exceed 25 μg/L during the period of June 1 to October 31, and temperature from Nine Mile Bridge (RM 58.0) to the Idaho border (RM 96.0) and Long Lake reservoir shall not exceed 20°C due to human activities

(Cusimano, 2003). The dissolved oxygen criterion for Long Lake is 'no measurable change from natural conditions,' while the minimum criterion for the river (from Nine Mile Bridge to the Idaho border) is 8.0 mg/L, which is to apply at all times (Cusimano, 2002). However, existing water quality does not always meet these criteria. Ambient data at the Stateline suggest that water temperature in the river and tributaries during the summer period exceed the criterion of 20°C (Cusimano, 2003; Cusimano, 2004). Dissolved oxygen concentrations caused by periphyton respiration during late summer (Cusimano, 2003) and diurnal minimums have been predicted to violate the river water quality criterion (Cusimano, 2002). Increasing nutrient level trends in the river under growing population of the Spokane Valley and increasing wastewater discharges have resulted in violation of Washington State water quality standards at several portions of the river, and are listed in one or more of Washington State Department of Ecology's 1996, 1998, 2004, 2006 and 2008 lists of impaired water bodies (Moore and Ross, 2010). TMDLs have been established in the Spokane River for phosphorus to control excessive algae blooms that contribute to both an aesthetic problem and a low dissolved oxygen level during low flow in the summer months. Results from the long term water quality simulation of the Spokane River-Long Lake system in this study were compared to low flow analysis to investigate the nutrient reduction requirements bearing in mind the existing water quality criterions.

2.3 Spokane River Water Quality: Management Efforts and Existing Studies

Degraded water quality in the Spokane River was first detected during 1920s. In 1889, the City of Spokane built a sewage system that dumped raw sewage directly into the river, which was visibly noticeable by 1920. In 1957 a primary treatment facility was installed. However, it was soon deemed inadequate by the Washington State Department of Ecology (Ecology), which led to the construction of a more advanced treatment plant in 1975 (Edmondson, 1996). Nonetheless, with increasing development in the area, water quality in the Spokane River kept degrading, and proper management became a necessity. Before long, Ecology was required, by court orders, to determine the maximum permissible phosphorous loading from all sources to the Spokane River for protecting beneficial uses in the Long Lake. Consequently, the Spokane River wasteload allocation process began in 1979.

Singleton (1981) prepared a supplemental report quantifying the levels of phosphorus and related parameters in the river system, and identified the sources of phosphorus contributing to the system and the deleterious effects of the high phosphorus levels. This report later served as a basis for the subsequent studies on the Spokane River wasteload allocation. A report by Funk et al. (1983) guided in establishing information for water quality management of the Spokane River during the early 1980s. The study characterized patterns for the water quality parameters, including temperature, dissolved oxygen, nitrogen, phosphorous, suspended solids, and BOD. Funk et al. (1983) reported that the Spokane Aquifer had a considerable impact on river temperature and nitrate-nitrogen concentrations. The study also conducted limited research on the effect of effluent on primary and secondary producers in the Upper Spokane River.

Gibbons et al. (1984) performed a baseline study in 1984 to determine the water quality, and primary and secondary producers of the Spokane River, and found out that the physicochemical and biological water quality of the river decreased as the water moved downstream. They suspected the increase in human densities and activities along the river and the associated increase in urban runoff as the reason for the water quality degradation. This study also helped in the assessment of short-term changes in river water quality after the initial operation of the Liberty Lake Sewage Treatment Plant.

Patmont et al. (1985) performed a study during low flow season of 1984, and determined that more than 40% of the influent total phosphorous load from Lake Coeur d'Alene was lost within the river system during transport. They indicated that this attenuation was through biological uptake and adsorption on the river bottom. This information was incorporated into a predictive model of phosphorous transport through the river system (Patmont et al., 1985). Generally appropriate for a variety of phosphorous loading scenarios, the model was intended for allocating phosphorus wasteloads. Patmont et al. (1987) updated the Long Lake data base, which was later used in refining the existing water quality models at that time. Following this study, phosphorous standards for Long Lake were proposed. Evaluation of hypothetical wasteload allocation scenarios by Patmont et al. (1987) indicated that reduction in future point source nutrient discharges was necessary to achieve the proposed phosphorous standards.

In 1994, Pelletier (1994) developed a steady state QUAL2E model of dissolved oxygen in the Spokane River from river mile 83.0 to 72.8. The study found the reach between Inland Empire Paper Company (IEPC) and Upriver Dam to be the most sensitive to dissolved oxygen changes due to BOD loading from IEPC. The QUAL2E model was then used to determine the wasteload allocations for BOD loading from IEPC to protect DO standards in the river. Pelletier (1997) later revised the wasteload allocations for BOD loading from IEPC based on new data collected by Inland Empire Paper Company.

Separate wasteload allocations were estimated for July-September and October-June to account for the variations in loading capacity from seasonal changes in river flow, temperature and DO. The revised wasteload allocations were found to be similar to, but more restrictive than, the wasteload allocations previously proposed by Ecology.

As the major source of recharge to the Spokane River comes from Coeur d'Alene Lake in Idaho, water quality of Coeur d'Alene Lake has significant impacts on Spokane River water quality. With the view to addressing water quality problems in Lake Coeur d'Alene (Idaho), Woods (1989) published a report on the hypolimnetic concentrations of dissolved oxygen, nutrients, and trace elements in the lake. Later in 1999, Idaho Department of Environmental Quality (DEQ) performed an assessment study of Coeur d'Alene Lake and proposed maximum daily loads for temperature and sediment (Idaho DEQ, 1999). In 2011, Idaho DEQ collected new bathymetry data and updated their assessment for accurate phosphorus loading calculations (Idaho DEQ, 2013).

With continuing efforts to improve water quality in the Spokane River, the 'Spokane River and Lake Spokane (Long Lake) Pollutant Loading Assessment for Protecting Dissolved Oxygen' project was undertaken in 1999. As a part of the project, Ecology compiled historical data and conducted a series of water quality surveys in 1999 and 2000. In addition, the Spokane River Phosphorus Technical Advisory Committee discharger members collected data during the summer of 2001. Report by Cusimano (2003) provides a summary of these field measurements and chemical data. These data, along with flow, water level, meteorological, and bathymetry data from other sources, were used to develop and calibrate a hydrodynamic and water quality model (CE-QUAL-W2) of the Upper Spokane River system by Berger et al. (2002). The model was intended to be used by Ecology to make recommendations on the total maximum daily load pollutant limitations for the Spokane River and Lake Spokane. Cusimano (2004) performed the 'Spokane River and Lake Spokane (Long Lake) Pollutant Loading Assessment for Protecting Dissolved Oxygen' study to assess the impacts of point and nonpoint sources of pollutants on dissolved oxygen concentrations. The CE-QUAL-W2 modeling results indicated that dissolved oxygen water quality criteria were violated during critical conditions in some areas of the Spokane River and Lake Spokane, and that the current loading of organic material and nutrients from both point and nonpoint sources was required to be reduced to meet the allowable concentrations. The study was also used to evaluate the existing total phosphorus criterion and associated total daily maximum load (TMDL) for Lake Spokane. Hydrodynamic and water quality models of the Spokane River and Lake Roosevelt system were later linked together in 2009 to develop phosphorus TMDL for the entire system (Berger et al., 2009).

Prior to mid-2000s, controlling point sources of pollution was the priority for Ecology. However, Washington still suffered from water quality degradation even after controlling majority of point source discharges (Hashim and Bresler, 2005). Nonpoint sources of pollution were then identified as the cause of these water quality problems. In June 2005, Washington proposed a water quality management plan to control nonpoint sources of pollution (Hashim and Bresler, 2005). The latest regulations of management plan to control nonpoint sources of pollution in Washington can be found in Rau (2015).

With continual efforts to maintain DO levels in the Spokane River-Long Lake system, the 'Spokane River and Lake Spokane Dissolved Oxygen TMDL' report in September, 2007 (Moore and Ross, 2007) established limits for ammonia (NH_3), total phosphorus (TP) and carbonaceous biochemical oxygen demand (CBOD) in the Spokane River. The TMDL focused on strategies to reduce phosphorus, because these strategies will likely result in reductions of ammonia and carbonaceous biochemical oxygen demand as well. 'Foundational Concepts for the Spokane River TMDL Managed Implementation Plan' was developed by the Spokane River TMDL Collaboration to help guide the implementation of this TMDL over the next 20 years. Phosphorus targets were expected to be achieved by installing the most effective feasible phosphorus removal treatment technology.

Moore and Ross (2010) later published a revised version of the 'Spokane River and Lake Spokane Dissolved Oxygen TMDL' report that identified dissolved oxygen responsibilities for hydroelectric dam operations in Lake Spokane, and reported that phosphorus is the nutrient that has the greatest effect on dissolved oxygen levels in the system. In addition to installing advanced wastewater treatment technologies, wastewater treatment plants were further required to reduce nutrients through actions such as obtaining offsets from nonpoint source reductions, water conservation, and wastewater reuse in order to achieve 90% reduction in total phosphorus discharge during the critical period, which would create a 63% decrease in total phosphorous in the river (Ecology, 2012). In addition to wasteload allocation for ammonia and carbonaceous BOD, wasteload allocations were established for stormwater discharges from municipalities with stormwater discharge permits. Load allocations for total phosphorus, ammonia, and carbonaceous BOD were assigned to the mouths of the main three tributaries to the Spokane River, and phosphorus allocations were assigned to groundwater discharges to the river and for groundwater and surface water to the Lake Spokane watershed (Moore and Ross, 2010).

2.4 Climate Change Studies on Spokane River

Several studies have been conducted to explore the impacts of climate change on the Spokane River Watershed (Fu et al., 2007a; Fu et al., 2007b; Barber et al., 2011; Jin and Sridhar, 2012). Fu et al. (2007a) developed and implemented a methodology to estimate the impacts of global climate change on the Spokane River Watershed using ArcGIS Geostatistical Analyst, and found that in the Spokane River Basin, increased precipitation consistently increases annual streamflow, while increased temperature consistently decreases annual streamflow because of decreased snowpack. Fu et al. (2007b) extended single parameter precipitation elasticity of streamflow index into a two parameter climate elasticity index, as a function of both precipitation and temperature, in order to assess climatic effects on annual streamflow for the Spokane River. In another study, Jin and Sridhar (2012) used the SWAT model to perform a study on the impacts of climate change on hydrology and water resources in Spokane River basins.

Keeping in mind the issue of aquifer pumping and climate change impacts on summer low flows in the Spokane River, Barber et al. (2011) developed a MODFLOW model to assess aquifer recharge and natural recovery feasibility study for the Spokane Valley-Rathdrum Prairie (SVRP). Results showed that increases in head by artificial recharge produce increased flows into gaining reaches and decreased flow out of losing reaches.

Climate projections in the Spokane area between 2010 and 2060 predict a 0.1 to 3.5°C warming and a -6.7 to 17.9% precipitation change (Jin and Stridhar, 2012). For the Spokane River Basin, hydrologic models predict increased runoff in the fall, decreased runoff in the spring, and consistent to slightly increased flow in the summer (Jin and

Stridhar, 2012). On the contrary, low summer flows have been decreasing rapidly in the Spokane River for somewhat unclear reasons (Barber et al., 2009). This does not imply that the climate hydrologic predictions are wrong; rather, climate effects might not be telling the whole story on low summer flows in the Spokane River.

2.5 Model Background, Capabilities, Limitations, History, and Selection

Historically 1-dimensional, steady state models (e.g., QUAL2K) have been used in water quality modeling (Bowie et al., 1985; Tillman, 1992; Van Orden and Uchrin, 1993; Tsihrintzis et al., 1995). However with the changing needs of time, more advanced, comprehensive and versatile stream water quality models should be implemented that can simulate the major reactions of nutrient cycles, algal production, benthic and carbonaceous demand, atmospheric reaeration and their effects on the dissolved oxygen balance. CE-QUAL-W2, developed by the U.S. Army Corps of Engineers, is a two-dimensional, laterally averaged, hydrodynamic water quality model capable of handling these major chemical and biological processes (Annear et al., 2001). The following sections discuss the model's background and rationale for selection of CE-QUAL-W2 model for this study, model capacity and limitations, and previous applications.

2.5.1 Background of CE-QUAL-W2

Originally known as the Laterally Averaged Reservoir Model (LARM), CE-QUAL-W2 is a finite difference model which uses grids to numerically solve the governing equations (Cole and Wells, 2003). The computed values (temperature, concentration, etc.) for each cell in CE-QUAL-W2 model are constant across the width of that cell. The model has been under continuous development since 1975. Water quality algorithms were added in 1986 by the Water Quality Modeling Group at the US Army Engineer Waterways Experiment Station (WES) and the model was renamed CE-QUAL-W2 Version 1.0. CE-QUAL-W2 has evolved from that time to include new algorithms to improve accuracy and stability (Cole and Wells, 2002). To date, the model has been successfully applied to many rivers, lakes, reservoirs, and estuaries (Annear et al., 2001). Further information on CE-QUAL-W2 model development history, application history, model capabilities and limitations is available at http:/www.ce.pdx.edu/w2.

2.5.2 CE-QUAL-W2 Capacity and Limitations

In general, CE-QUAL-W2 can handle a branched and/or looped system with flow and/or head boundary conditions. CE-QUAL-W2 model allows the user to use the quickest numerical scheme for constituent transport (Annear et al., 2001). In addition to temperature, CE-QUAL-W2 can simulate as many as 20 other water quality variables (Annear et al., 2001). The model can handle major chemical and biological processes like effects of DO on atmospheric exchange, photosynthesis, respiration, organic matter decomposition, nitrification, and chemical oxidation of reduced substances; uptake, excretion, and regeneration of phosphorus and nitrogen and nitrification-denitrification under aerobic and anaerobic conditions; carbon cycling and alkalinity-pH-CO_2 interactions; trophic relationships for total phytoplankton; accumulation and decomposition of detritus and organic sediment; and coliform bacteria mortality (Annear et al., 2001).

Other models developed for river basin modeling (WQRSS, HEC-5Q, and HSPF) have serious limitations in terms of their ability to resolve the vertical and longitudinal circulation patterns within a reservoir (Annear et al., 2001). CE-QUAL-W2 has the ability to model both longitudinal and vertical gradients in water quality, which overcomes the restriction of above-mentioned model's inability to compute 2-D circulation within reservoir systems (Annear et al., 2001). Another primary advantage of CE-QUAL-W2 is that the Manning's friction factor did not need to be varied as the river stage increased (Wells, 1999). Version 3.1 allows for modeling of multiple algal groups and output of kinetic fluxes for multiple constituents including phosphorus (Cole and Wells, 2003). Moreover, CE-QUAL-W2 River Basin Model, Version 3.1 was proposed as the most appropriate for modeling the Spokane River-Long Lake River Basin by Annear et al. (2001) because of the following elements:

- capability of replicating density stratified environment
- availability of modeling multiple CBOD groups
- the scope of extending the model to the entire Spokane basin
- capability of accurately representing hydraulic elements of dams
- ability of seamless linkage between the river and reservoir.

The limitation of the CE-QUAL-W2 model which may affect this study is that it does not explicitly include zooplankton and their effects on recycling of nutrients (Cole and Wells, 2003). The model also uses a simplistic algorithm to simulate sediment oxygen demand. "It does not model kinetics in the sediment and at the sediment-water interface.

This places a limitation on long-term predictive capabilities of the water quality portion of the model" (Cole and Wells, 2002). Despite these limitations CE-QUAL-W2 has accurately modeled the behavior of many water bodies (Cole and Wells, 2002).

2.5.3 Previous CE-QUAL-W2 Modeling Efforts

CE-QUAL-W2 has been a popular model for various water quality models that include turbidity, dissolved oxygen, eutrophication, effects of macrophytes, and more. To date, the number of waterbodies modeled by CE-QUAL-W2 is more than 250 (Nielsen, 2005). CE-QUAL-W2 has been used to simulate water quality fluctuations of reservoirs (Cole and Tillman, 1999; Bunch et al., 2003; Singleton et al., 2013; Haddad et al., 2015; Chang et al., 2015). Studies that have used CE-QUAL-W2 to simulate the hydrodynamics and water temperature of rivers, reservoirs, lakes include Cole and Tillman (1999), Smith et al. (2012), Buccola et al., (2013), and Ma et al. (2015). Spatial and temporal variations in dissolved oxygen concentrations and other water quality variables have been predicted using CE-QUAL-W2 by Martin (1988), McIntyre et al. (2003), Berger and Wells (2008), and Smith et al. (2012). Water quality modeling using CE-QUAL-W2 has been done for eutrophication by Cerco and Cole (1993), Kuo et al. (2006), Ha and Lee (2008), and Zhang et al. (2008). Effects of macrophytes on hydrodynamics and water quality were modeled by Berger and Wells (2008) using CE-QUAL-W2.

Phytoplankton dynamics in rivers, lakes and reservoirs have also been modeled using CE-QUAL-W2 by McIntyre et al. (2003), Berger and Wells (2008), and Ma et al. (2015). Jeznach and Tobiason (2015) used CE-QUAL-W2 to simulate the effects of increasing future air temperatures on water temperature, stratification timing and duration.

CE-QUAL-W2 has been used for modeling of algal succession and nutrient dynamics with the inclusion of multiple algal groups (Flowers, 2001; Cusimano, 2003; Nielsen, 2005). Turbidity has been widely modeling using CE-QUAL-W2 (Gelda and Effler, 2007; Gelda et al., 2012; Samal et al., 2013).

CE-QUAL-W2 has also been implemented to model fecal coliforms (Hammond, 2004), water column stability (Colarusso et al., 2003), thermal stratification and salinity (Ebrahimi et al., 2015), algal community dynamics (Smith et al., 2014), best management practices (Wu et al., 2006; Sullivan et al., 2013), density currents (Chung and Gu, 1998; Ma et al., 2015), estuarine water quality (Bowen and Hieronymus, 2003), freshwater mussels (Zhang et al., 2008), lake water quality (Williams, 2007; McCulloch, 2011), and effects of dam removal (Perry et al., 2011).

Zhang et al. (2015) recently improved the CE-QUAL-W2 model by integrating the benthic sediment diagenesis module into the model, making the model capable of simulating the dynamic releases of ammonium, nitrate, phosphorus, dissolved silica and dissolved methane from the sediment to the overlying water. Hanna (2014) also made efforts to implement sediment transport model for CE-QUAL-W2. Rounds and Buccola (2015) attempted to enhance and augment new features in CE-QUAL-W2 to help dam operators and managers explore and optimize potential solutions for temperature management downstream of thermally stratified reservoirs.

2.5.3.1 Development of Spokane River CE-QUAL-W2 Model

CE-QUAL-W2 model development for the Spokane River has been documented in several reports. CE-QUAL-W2 Version 2 was used for Long Lake initially, but the model was not able to simulate the river sections (Annear et al., 2001), which led to the development of Version 3.1. Annear et al. (2001) describe the boundary conditions and model setup for the Upper Spokane River Model for year 1991 and 2000. This model simulates the water quality of the river reach from the Stateline with Idaho (River mile 96.0) to Long Lake dam (RM 32.5).

Annear et al. (2001) summarize the background data used in the modeling effort, including the inflows, temperatures, and water quality of river and tributaries, meteorological conditions, bathymetry of the Spokane River, dam pools along the river, point source inflows and water quality characteristics, reservoir operations and structure information. The report also discusses why CE-QUAL-W2 Version 3.1 was selected for the modeling of Spokane River. Evaluation of the 1991 and 2000 model calibration and discussion issues relative to the calibration effort can be found in Berger et al. (2002).

With the availability of considerable field data from City of Spokane and other point dischargers to the Spokane River during 2001, the City of Spokane funded to continue developing the Spokane River model for the year 2001 (Slominski et al., 2002). Similar to Annear et al. (2001), Slominski et al. (2002) describe the boundary conditions and model setup for 2001. The calibration effort of the 2001 Upper Spokane River Model, focusing on model predictions of hydrodynamics, temperature, and eutrophication model parameters, is available in report by Berger et al. (2003).

Refinements were later made by Ecology and Portland State University to the Spokane River model calibration since the original calibration of the model, the details of which can be found in Berger et al. (2004).

2.5.4 Model Selection

Primarily, the CE-QUAL-W2 model is appropriate for the Spokane River-Long Lake system, as it is a long and narrow reservoir, where changing concentrations across the width of the reservoir can be assumed to be insignificant. The model also allows for long term simulations and water quality responses which are important for this study. Because of its extensive capabilities and the significant amount of work previously completed using this model, CE-QUAL-W2 Version 3.1 was selected to model the water quality in the Spokane River-Long Lake system.

CHAPTER 3

DATA COLLECTION AND MODEL DEVELOPMENT

3.1 Overview of Chapter 3

Waste Load Allocations from point and nonpoint source loadings in the United States have traditionally been based on historic hydrologic and water quality data. Total Maximum Daily Load (TMDL) policies and Municipal Separate Storm Sewer System (MS4) permits are often based on steady-state low flow analysis with little thought about unsteady flow and pollutant loading conditions prevalent during stormwater runoff events. The overarching goal of this research is to improve our understanding of how changes in watershed conditions due to population growth and climate change may alter management decisions with respect to both wasteload allocation and load allocation in the TMDL process, using the Spokane River watershed as a case study. As discussed in the previous chapter, the two-dimensional, laterally averaged, numerical model CE-QUAL-W2 was selected for simulation of the hydrodynamics and water quality of Spokane River and Long Lake. Specifically, the model will be used to assess the impact of climate and population change scenarios on selected water quality constituents related to nutrients and dissolved oxygen levels. The following sections describes the data collection and model development process for the Spokane River CE-QUAL-W2 model.

3.2 Spokane River-Long Lake Bathymetry, and Model Grids

Bathymetry is the measurement of depths of a body of water and describes the shape and volume of that waterbody. It quantifies the height, length, width and orientation of each cell used in a grid to describe the reservoir/river. The bathymetry of the 39.2 miles of the Spokane River between the Washington-Idaho Stateline and Long Lake and Long Lake itself developed by Washington State Department of Ecology (Ecology) has been used in this study.

Details on the development of bathymetry of the Spokane River-Long Lake system from Digital Elevation Model, GIS coordinates, cross sections, and contour maps can be obtained from Annear et al. (2001) and Slominski et al. (2002). The bathymetry of the Spokane River and Long Lake was then used by Ecology to develop the model grid. The decision of the location of the break in the branches and water-bodies was based on (1) how groundwater inflow/recharge was computed for the river, (2) how the vertical slope changed from branch to branch, and (3) location of pools or dams (Annear et al., 2001). The model complexity in the vicinity of Monroe Street Dam and Upper Falls Dam was simplified in the Ecology model by explicitly avoiding modeling of the Monroe Street Dam (Annear et al., 2001). After passing through the dam, water is placed in the model river segment downstream of the Upper Falls Dam, neglecting the Monroe Street Dam.

Final model resulted in 6 waterbodies containing a total of 12 branches, represented by 189 segments each having 47 vertical layers. The branch layout, specified by these parameters, is discussed in the “Upper Spokane River Model: Boundary Conditions and Model Setup, 1991 and 2000” report (Annear et al., 2001). The grid layout, adapted from Annear et al. (2001), is shown in Figure 3.1.

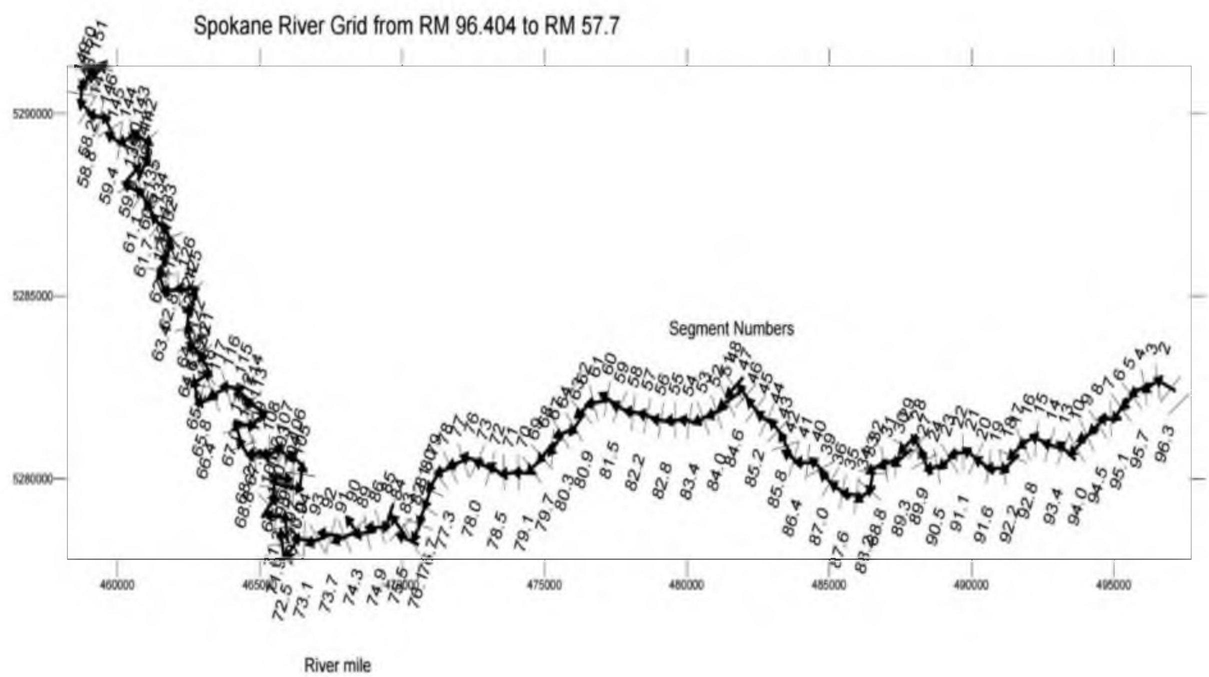


Figure 3.1 Plan View Spokane River Grid (Arrows Show the Segment Orientation)

Table 3.1 and Table 3.2 show the water-bodies layout and branches layout (Annear et al., 2001) for the Spokane River model. The vertical layout of the Spokane River grids, upstream of Long Lake, is shown in Figure 3.2 (adapted from Annear et al., 2001). The x-axis shows the river miles, while elevations are placed on the y-axis. The figure includes the locations of the branches, highest (red dots) and lowest (blue triangles) elevations recorded in a cross-section of the river, the water surface elevation (WSE) from a GIS map (blue line), and a light blue line showing the elevations of the dam spillways or pools. Annear et al. (2001) contains detailed discussion on the vertical layout of Spokane River grids.

Table 3.1 Water Body-Branch Layout

Water body (WB)	**Branch Start**	**Branch End**
WB 1: 4 sloping branches above the pool of Upriver Dam	1	4
WB 2: Pool of Upriver Dam	5	5
WB 3: Pool of Upper Falls Dam	6	7
WB 4: 2 sloping branches above the Nine Mile Dam pool	8	9
WB 5: Nine mile Dam pool	10	11
WB 6: Long Lake pool	12	12

Table 3.2 Layout of Branches for the Spokane River and Long Lake

Branch #	Location	Start RM	End RM	Length	# of Segments	DLX, m	Bottom Elev. Start	Bottom Elev. End
	Post Falls USGS gage to Stateline							
1	Stateline to Harvard Road Bridge	96.40	93.82	4145.52	9	461.61	616	608.5
2	Harvard Road Bridge to Barker Road Bridge	93.82	90.34	5608.39	12	467.37	608.5	600
3	Barker Road Bridge to RM 87.50	90.34	87.50	4570.59	10	457.06	600	585
4	RM 87.50 to The Islands Foot Bridge	87.50	84.45	4916.24	10	491.62	858	578
5	The Islands Foot Bridge to Upriver Dam	84.45	80.18	6857.87	14	489.85	571	571
6	Upriver Dam to Green Street Bridge	80.18	78.10	3352.30	7	478.90	560	560
7	Green Street Bridge to Upper Falls Dam	78.10	74.75	5396.20	11	490.56	560	560
8	Upper Falls Dam to Spokane USGS gage	74.75	72.93	2925.69	6	487.62	525	517.5
9	Spokane USGS gage to Seven Mile	72.93	63.20	15659.24	32	489.35	517.5	485
10	Seven Mile to RM61.813	63.20	61.81	2232.19	5	446.44	481	481
11	RM 61.813 to Nine Mile Dam	61.81	57.77	6506.66	14	464.76	481	481
12	Nine Mile Dam to Long Lake Dam							

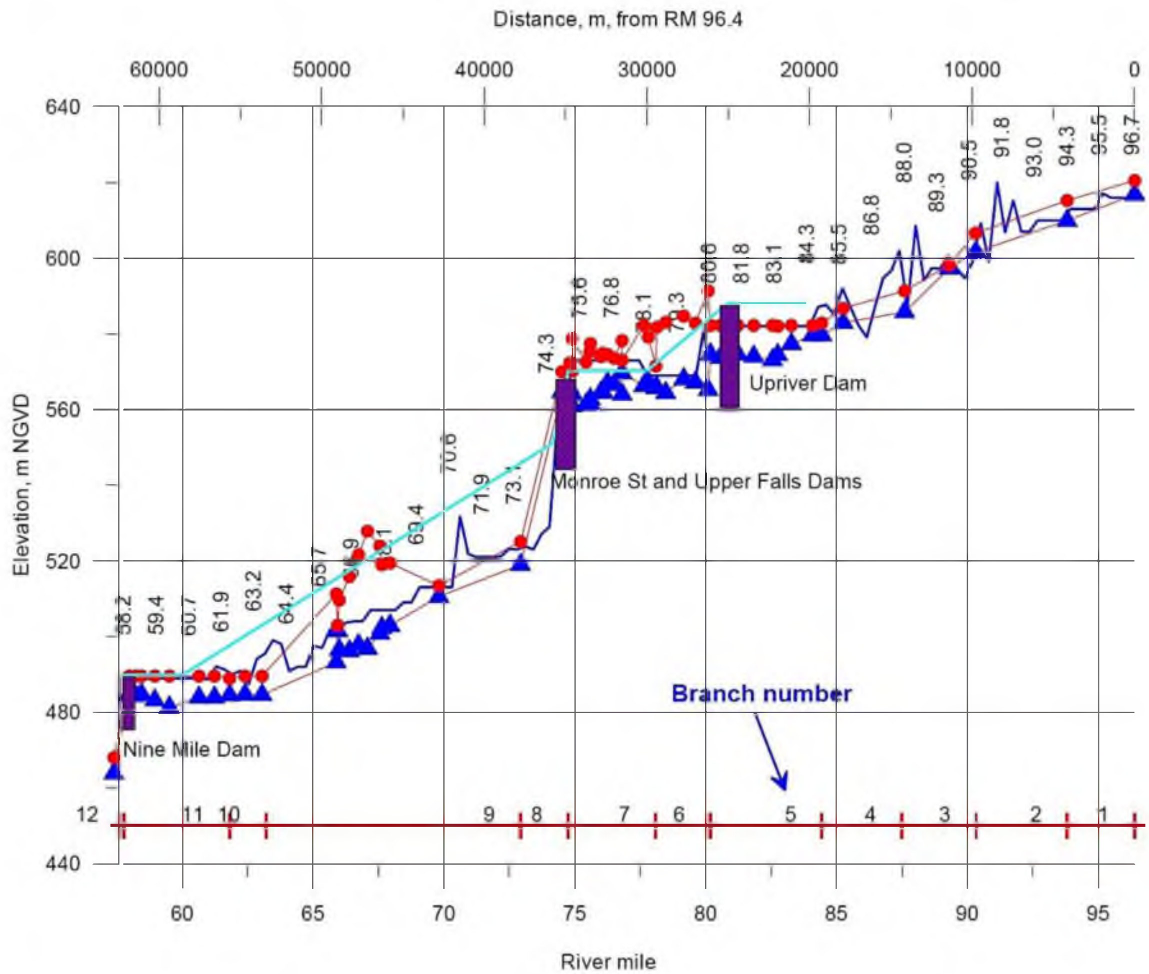


Figure 3.2 Vertical Layout of Spokane River Grid (Adapted from Annear et al., 2001)

3.2.1 Reduction of Vertical Cells

The CE-QUAL-W2 Spokane River Model (version 3.1) for 2001 developed by Ecology runs from Julian day 74 (March 15) to Julian day 304 (October 31), which takes about 18 hours for a complete model simulation. While the run time is reasonable for a single year, it was not appropriate for long term simulations. Therefore, the Ecology model needed to be simplified so that the simulation could be completed in less time. One way to reduce simulation time is to reduce the number of active cells (horizontal, vertical).

Changing the number of horizontal cells (essentially the model segments) would require redeveloping the bathymetry. Reducing the number of vertical cells in each segment seemed feasible and acceptable as long as it did not affect the model results significantly.

The change was made in the bathymetry files, where the number of vertical cells were reduced from 47 to 24. Two cells were combined to form one cell (for example: combining cell number 2 and 3 formed new cell number 2, combining cell number 4 and 5 formed new cell number 3, and so on). This change was made for every segment in each bathymetry file. The thickness of each layer was also changed by doubling the layer thickness so that the total depth of the segments remain unchanged. The topmost and bottommost cells were inactive cells, which were kept unchanged with zero values for all entries. The width of the new cells were obtained by taking average width of the two combined cells. It was confirmed that there were 24 values for cell depth, and 24 corresponding values for cell width for every segment in the bathymetry files. It was important to make sure that the entries in the bathymetry files were arranged in an orderly manner to avoid any disruption in model simulation. The length of segments, water surface elevation, orientation of segments (in radian), and manning's n were kept unchanged from the Ecology Model.

The modified model was then simulated for the same time period as the Ecology model (Julian day 74-304, total 230 days). Reducing the number of vertical cells by half resulted in decrease of simulation time by nearly 50%, while maintaining nearly identical model results. The modified model with reduced vertical cells was therefore considered acceptable and used for further simulations. Results for Long Lake with the modified Spokane River Model are compared with original Ecology model in Figure 3.3.

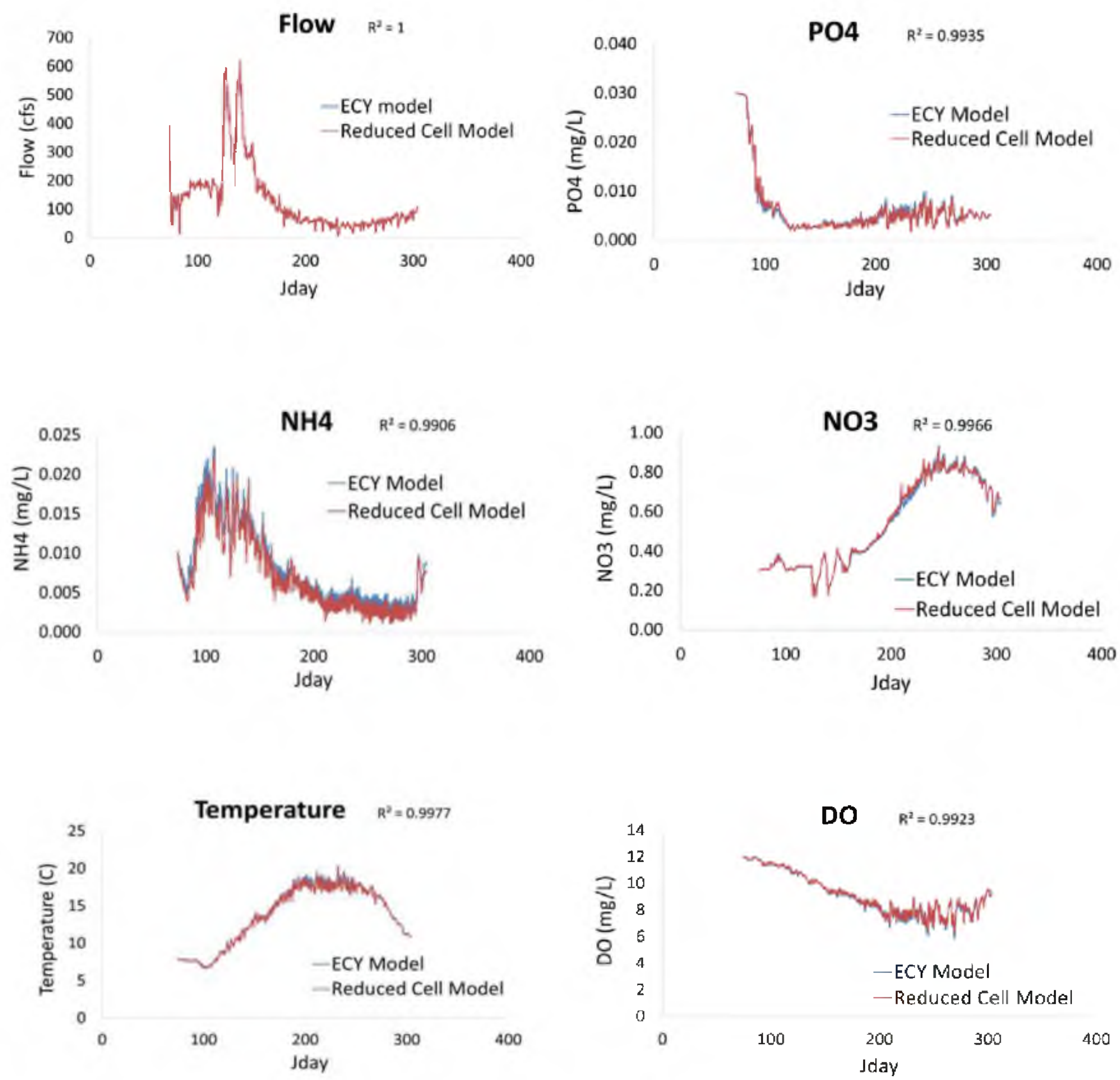


Figure 3.3 Comparing 2001 Results (Long Lake, Segment 188) of Modified Model (Reduced Vertical Cells) and Ecology (ECY) Model

3.3 Initial Condition

Each simulated constituent must have an initial, single concentration for the entire model or a grid-wide initial vertical profile of concentrations at the start of each model run. Initial constituent concentrations were considered uniform throughout the river for every segment and layer. These values include phosphate-phosphorous, ammonia-nitrogen, nitrate-nitrogen, labile dissolved organic matter (LDOM), refractory dissolved organic matter (RDOM), labile particulate organic matter (LPOM), refractory particulate organic matter (RPOM), algae groups, dissolved oxygen, total inorganic carbon (TIC), alkalinity, total dissolved solid (TDS), tracer, coliform, conductivity, chlorine, inorganic suspended solid (ISS), dissolved and particulate silicon, total iron, and CBOD groups.

Ecology estimated the initial values of these parameters based on the long-term averages at various sampling locations within the study area; the values are shown in Table 3.3. Initial value for other parameters – TDS, tracer, coliform, conductivity, chlorine, ISS, dissolved and particulate silicon, total iron, and all CBOD groups – were set to zero in the model. Parameters like tracer, coliform, conductivity, chlorine, ISS, TIC, TDS, alkalinity, silicon, and total iron were not central to the objectives of this study, but needed to be used to complete model input requirements.

3.4 Boundary Condition

Accurate boundary conditions are essential to properly represent any process with computer models. Incomplete or faulty boundary conditions are often the source of error found in modeling efforts. In the Spokane River model, the Washington-Idaho Stateline

Table 3.3 Initial Condition for Model Parameters

Parameter	Initial Value (mg/L)					
	Branch 1	**Branch 2**	**Branch 3**	**Branch 4**	**Branch 5**	**Branch 6**
PO_4-P	0.03	0.03	0.03	0.03	0.03	0.03
NH_4-N	0.01	0.01	0.01	0.01	0.01	0.01
NO_3-N	0.30	0.30	0.30	0.30	0.30	0.30
PO_4-P	0.03	0.03	0.03	0.03	0.03	0.03
NH_4-N	0.01	0.01	0.01	0.01	0.01	0.01
NO_3-N	0.30	0.30	0.30	0.30	0.30	0.30
LDOM	0.10	0.10	0.10	0.10	0.10	0.10
RDOM	0.10	0.10	0.10	0.10	0.10	1.00
LPOM	0.10	0.10	0.10	0.10	0.10	0.10
RPOM	0.10	0.10	0.10	0.10	0.10	1.00
ALG 1	0.10	0.10	0.10	0.10	0.10	0.10
ALG 2	0.10	0.10	0.10	0.10	0.10	0.10
ALG 3	0.10	0.10	0.10	0.10	0.10	0.10
DO	12.0	12.0	12.0	12.0	12.0	12.0
TIC	5.0	5.0	5.0	5.0	5.0	5.0
ALK	19.8	19.8	19.8	19.8	19.8	19.8

was considered as the upstream boundary. The upstream boundary conditions included Stateline inflow estimated from USGS flow station and controlled releases from Post Falls Dam (Idaho), water temperature measurements, and water quality concentrations obtained from water quality monitoring program of Ecology. The upstream boundary conditions can be adjusted; however observed values were used in the model.

Boundary condition also included Hangman Creek, Little Spokane River, Coulee/Deep Creeks, and groundwater contributions. Hangman Creek, Coulee/Deep Creeks, and Little Spokane River having drainage areas of 689 square mile, 543 square mile, and 700 square mile, respectively, enter the Spokane River in the model at segments 95, 145, and 151 at River Mile (RM) 72.4, 56.4, and 58.8. Similar to the upstream boundary, Hangman Creek and Little Spokane River boundary condition was represented by USGS measured inflows, and Ecology monitored water temperature measurements and water quality concentrations. Groundwater contributions for each branch were represented in the model by inflows, water temperature, and water quality concentrations mostly estimated by Ecology.

The boundary conditions (BC) values are controlled in the model through separate input file for each location. The following sections discuss how the observed values were obtained or estimated to complete the boundary condition requirements of the model.

3.5 Data Collection

One of the tasks of this study was to modify Ecology's Spokane River CE-QUAL-W2 model to simulate hydrology and water quality over an 11-year historic period (1999-2009), with particular attention to phosphorus, nitrogen, dissolved oxygen, and river temperature. This required modifying each input file with data values from 1999-2009. The input data for the extended time period were used at intervals obtained from sources mentioned above. The model used internal interpolation to fill in the boundary conditions between the data. In some cases (described later) there were not much data available to characterize water quality constituents. The result is that those water quality constituents remained constant over time. This assumption did not have much influence on the water quality calibration due the small magnitude of their inflows. The following sections contain details on the data collection of each input parameters.

3.5.1 Temperature and Inflow Constituent Concentration

Inflow constituent parameters required by CE-QUAL-W2 included total dissolved solid (TDS), tracer, coliform, conductivity, chloride, inorganic suspended solid (ISS), PO_4-P, NH_4-N, NO_3-N, LDOM, RDOM, LPOM, RPOM, 10 CBOD groups, 3 algae groups, dissolved oxygen (DO), total inorganic carbon (TIC), and alkalinity. Water quality data were provided primarily by Ecology. Additional flow, temperature and water quality data were provided by the USGS in Washington and Idaho, dischargers along the Spokane River, and operators of the dam facilities. The data collected at these sites consisted of periodic grab samples, which were also used to generate longitudinal profiles (later used for model calibration) of the water quality parameters. Information on data stations used in this study is provided in Table 3.4.

Not all concentration values required by the model were present in the webpages. When data for a particular constituent were unavailable, concentrations were estimated from other relevant available data and literature.

Table 3.4 Station Number and Name of the Ecology Sites Used

Name of Station	Water body ID	Location, Elevation	River Mile
57A150 - Spokane River at Stateline Branch	WA-57-1010	Latitude 47.6985 Longitude 117.0446 Elevation 1980 ft.	96.35
55B070 - Little Spokane River near Mouth	WA-55-1010	Latitude 47.7829 Longitude 117.5305 Elevation 1525 ft.	1.1
56A070 - Hangman Creek at Mouth	WA-56-1010	Latitude 47.6546 Longitude 117.4543 Elevation 1720 ft.	0.6

Coliform, conductivity, TSS, phosphate-phosphorus, ammonia-nitrogen, nitrate-nitrogen, and dissolved oxygen data were obtained directly from the Ecology Data Center. Total dissolved solid, being related to conductivity, was obtained by multiplying the conductivity data by 0.65. Tracer values were kept at zero, as in the original Ecology CE-QUAL-W2 model (Ecology model).

The Ecology model had a constant chloride concentration value for the upstream boundary and tributaries. Similarly, the chloride concentration was kept constant in the extended model, with values unchanged from Ecology model. It was backed up by a sensitivity test by changing the chloride concentration in the Ecology model by ±10%, and running the model until completion. No difference in model results was obtained, indicating that chloride is not a sensitive parameter to the model results.

Ecology webpage had total suspended solids (TSS) data, but the model required data inputs of inorganic suspended solid (ISS). As there is no established method to obtain ISS from TSS, available TSS data were used in the model as a replacement of ISS. The rationale of the choice was that ISS could not be greater than TSS concentrations. To ensure that this assumption was adequate, the Ecology model was run with TSS concentrations instead of ISS. Very little (insignificant) difference was observed in model results, demonstrating ISS to be an inconsequential parameter for model results.

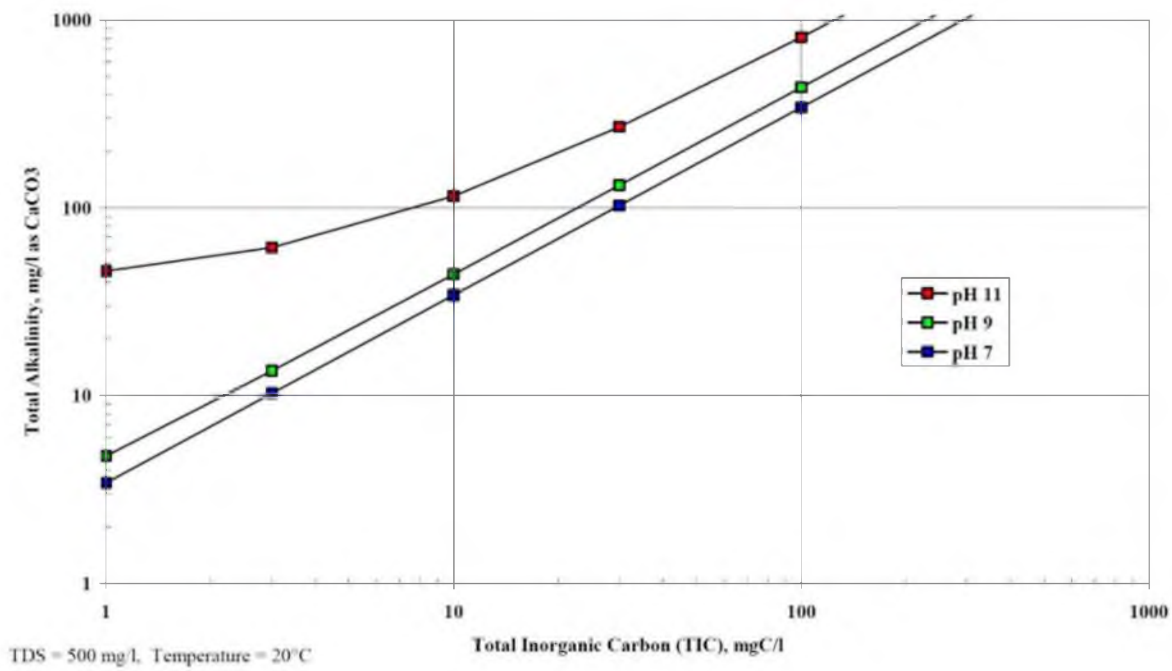
Organic matter concentrations at the upstream boundary condition, tributaries, and point sources were simulated for year 2001 by Ecology using CBOD ultimate data and multiple CBOD compartments in CE-QUAL-W2. The simulated CBOD concentrations for 2001 were assumed to be true for all years.

The constituent concentrations of LDOM (labile dissolved organic matter), RDOM (refractory dissolved organic matter), LPOM (labile particulate organic matter) and RPOM (refractory particulate organic matter) at upstream boundary and tributaries were set to zero, following the Ecology model. Algae groups 2 and 3 were represented by a constant concentration value in the Ecology model. The same was done for the extended model keeping algae concentrations unchanged from the Ecology model.

Algae concentrations for group 1 were primarily estimated for 2001 in the Ecology model using chlorophyll-a data, assuming a ratio of 35 μg/l chlorophyll-a to 1 mg/l algae (Annear et al., 2001; Slominski et al., 2002). However, chlorophyll-a data were unavailable for the other years. As a result, concentrations for algae group 1 for rest of the years were obtained from a multiple regression equation. Higher concentration of nutrients (PO_4-P, NH_4-N, and NO_3-N), higher temperature, and suitable pH can cause higher algae concentration.

A multiple regression equation was developed for each site using 2001 phosphate-phosphorus, ammonia-nitrogen, nitrate-nitrogen, dissolved oxygen, pH and temperature data obtained from Ecology and algae concentration from the Ecology model (R^2 = 0.73 - 0.95). The regression equation produced fairly good estimates of algae (group 1) concentrations for 2001, which was held true for rest of the years. More intensive algae level data compiled over longer time periods would possibly help in minimizing the discrepancies in estimating algal concentrations.

Total inorganic carbon (TIC) and alkalinity data (required by the model) were not present in the Ecology Data Center. Thus estimates had to be made. As pH is directly related to TIC, pH data from Ecology was used to estimate the TIC. Regression equation relating pH (from Ecology Data Center) and TIC (from Ecology model) was developed (R^2 = 0.86 - 0.96) for each site, which was used to obtain total inorganic carbon estimates for the entire period 1999-2009. To obtain alkalinity, a nomograph (Reference: Alkalinity Tech Note, http:// www.tocsystemsinc.com/products/alk/docs/alk%20tech%20note.pdf) relating total inorganic carbon, pH, TDS and temperature (shown in Figure 3.4) was used. Alkalinity is practically linear with total inorganic carbon at pH below 10. It is understood from literature that, for a constant temperature of 20°C and TDS of 500 mg/L, total inorganic carbon is related linearly to pH within a pH range of 7-9. As pH at Spokane River sites were within this range, the nomograph used produced good estimates of total inorganic carbon. Moreover, total inorganic carbon tends to increase with increasing temperature and TDS (Stumm and Morgan, 1981). As the temperature and TDS values at Spokane River sites were well below the conditions held for the nomograph, total inorganic carbon estimates obtained were considered the maximum possible at the given condition.



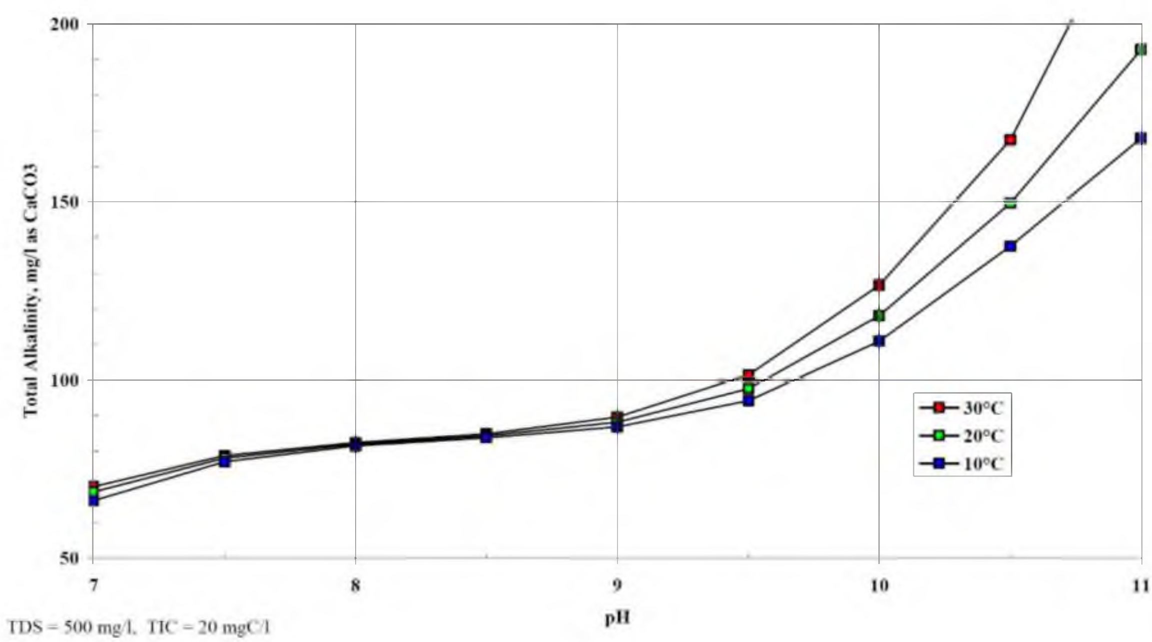


Figure 3.4 TIC Nomograph (Alkalinity Tech Note)

This assumption was backed up by the results of a sensitivity test with total inorganic carbon. The model was run until completion by changing the total inorganic carbon concentrations in the Ecology model by ±15%. No difference in results was obtained, indicating that total inorganic carbon is not a sensitive parameter to model results. Sensitivity tests were also performed for alkalinity, by changing the alkalinity concentrations in the Ecology model by ±20%. With no difference in model results, alkalinity was considered an insensitive parameter to the model results.

Since there is no active monitoring taking place on Coulee Creek, stream temperatures were unknown for 1999-2009. However, temperatures are monitored in the adjacent Hangman Creek basin. As Coulee Creek flows are much smaller than Hangman Creek, temperature and constituent concentrations at Coulee Creek were assumed to be equivalent to that of Hangman Creek. A report by GeoEngineers, Inc. (2011) reported some concentrations values at Coulee/Deep Creek during summer-2010, which was used wherever possible to maintain consistency of the water quality data. Figure 3.5 shows the observed water quality at Stateline, Hangman Creek, and Little Spokane River during 1999-2009.

Water temperatures showed a typical seasonal trend (high during summer, low duing winter). Water temperatures at Stateline were higher than Hangman Creek and Little Spokane River. Dissolved oxygen at Stateline varied between 6.9-13.8 mg/L. Nutrient concentrations were higher at Hangman Creek and the Little Spokane River compared to those measured at the Stateline. The highest concentrations typically occurred during January-April, which agree with the findings of Cusimano (2004). The loading of nutrients and organic material to the Spokane River from Hangman Creek mainly occurred during

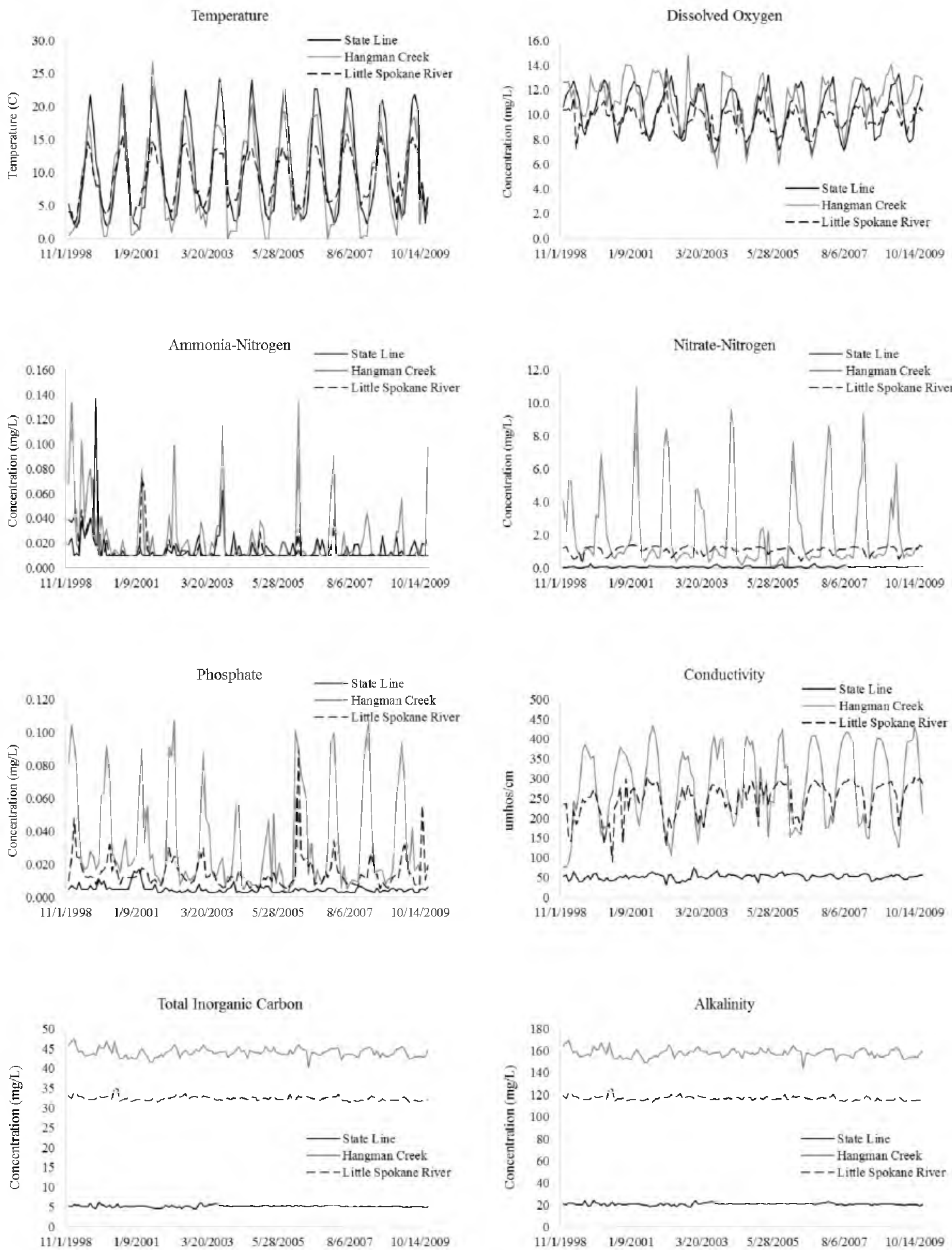


Figure 3.5 Boundary Condition Temperature and Constituent Concentration

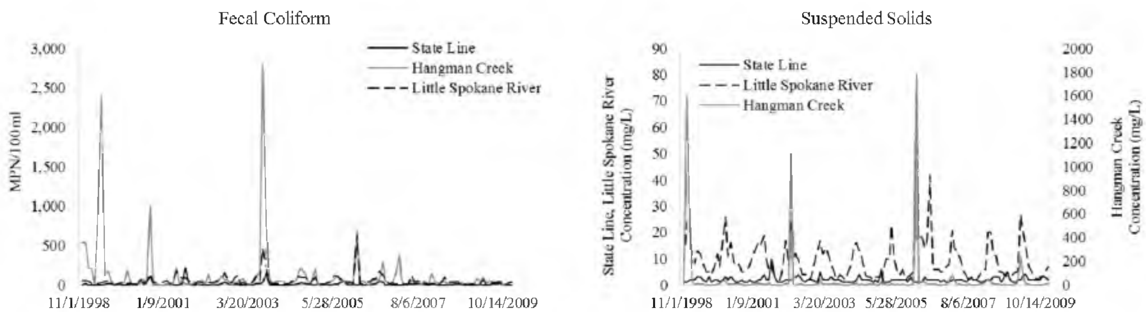


Figure 3.5 Continued

the late winter to the end of May, while loading from the creek during July-October was insignificant due to low discharge levels. On the other hand, loading from Little Spokane River during July-October was high relative to Hangman Creek because of the significant groundwater contributions (Cusimano, 2004).

Table 3.5 shows a summary of the water quality parameter concentrations used for boundary conditions. All units are in mg/L except for temperature, conductivity, and coliform which are in °C, umhos/cm, and MPN/100 ml units.

3.5.2 Flow at Spokane River Sites

The model required hourly flow data inputs. Observed flow records (cfs) at daily scale for upstream boundary and tributary sites were obtained from USGS database (see Table 3.6 for station details). The data were converted to hourly data (m^3/sec) as required by the model. Rather than assuming that flow changes linearly over the day, a smooth change in flow curve was maintained for better numerical stability of the model.

Table 3.5 Summary of Water Quality Parameter Concentrations

Variable	Stateline		Hangman Creek		Little Spokane River	
	Average	Range	Average	Range	Average	Range
Temperature	10.4	1.4 – 24.2	9.2	-0.2 – 26.8	9.1	2.2 – 16.2
DO	10.5	6.9 – 13.8	11.1	5.7 – 14.8	9.7	7.3 – 18.6
NH_4-N	0.015	0.010 0.137	0.024	0.010 –0.135	0.014	0.010 – 0.073
NO_3-N	0.07	0.01 – 0.26	2.15	0.09 – 11.0	1.10	0.28 – 1.41
PO_4-P	0.005	0.003 0.017	0.034	0.003 –0.109	0.015	0.005 – 0.090
Conductivity	53	32 – 75	283	77 – 433	249	90 – 307
TIC	5.3	4.5 – 6.3	44.1	40.2 – 47.4	32.5	31.6 – 35.1
Alkalinity	20.9	18.0 – 24.4	158.1	144.3 –169.8	117.0	113.9 – 126.4
Coliform	11	1 – 220	120	1 – 2800	51	3 – 640
Sus. Solids	2	1 – 10	48.1	1 – 1784	9.5	2 – 80

Table 3.6 USGS Station Name and Number

Name of Station	**Station ID**	**Location**
Hangman Creek at Spokane, WA	USGS 12424000	Latitude 47°39'10" Longitude 117°26'55" Elevation 1717.42 ft.
Little Spokane River at Dartford, WA	USGS 12431000	Latitude 47°47'05" Longitude 117°24'12" Elevation 1585.62 ft.
Spokane River Nr Post Falls, ID	USGS 12419000	Latitude 47°42'11" Longitude 116°58'40" Elevation 2050 ft.

Coulee Creek flows were not monitored during 1999-2009, and thus were estimated for the model. Adjacent to the Hangman Creek basin, flow estimates for Coulee Creek were made by comparing its basin area to Hangman Creek basin area and taking a fraction of the Hangman Creek flow.

$$Coulee\ Creek\ flow = \frac{Coulee\ Creek\ Basin\ area}{Hangman\ Creek\ Basin\ area} * Hangman\ Creek\ Flow$$

GeoEngineers, Inc. (2011) reported flow values at the Coulee/Deep Creek during summer, 2010 in the range 0-8.9 cfs, which was consistent with the estimates of 1999-2009 used in this study.

<u>3.5.2.1 Flow at Stateline</u>

The change in flow occurring between Post Falls and the Stateline was estimated by using flow data from Harvard Road (RM 93.7) and gage station near Post Falls, ID (USGS: 12419000). Flow rates at Harvard Road were typically less than those at Post Falls due to losses to the aquifer. The difference in flow between Post Falls and Harvard Road was then used to estimate the flow at the Stateline, which lay 4.7 miles downstream of Post Falls. When data were not available, flow rates were estimated by applying the following regression equation predicting flow at Harvard Road given data from Post Falls.

$$Q_{Harvard} = 0.00000199\ Q_{Post\ Falls}^2 + 0.9244\ Q_{Post\ Falls} - 68.8$$

The total distance between Post Falls and Harvard Bridge is 7.7 miles, and the loss/gain to the aquifer occurring between Post Falls and the Stateline was estimated by multiplying the difference in flow between Post Falls and Harvard Road by the fraction f of river miles between Post Falls and Stateline (f = 4.7 miles/7.7 miles). The gain/loss to the aquifer (typically a loss) between Post Falls and Stateline was estimated from

$$Q_{Aquifer} = (Q_{Harvard} - Q_{Post\ Falls})\frac{4.7}{7.7}$$

which was then used to estimate the flow at Stateline $Q_{Stateline}$ with

$$Q_{Stateline} = Q_{Aquifer} + Q_{Post\ Falls}$$

The average flows at Stateline, Hangman Creek, Little Spokane River, and Coulee Creek during this period were 165.2, 5.8, 8.1, and 0.73 m^3/sec, respectively. The peak flows occurred during winter/spring (February through April), because of the snowmelt runoff. Figure 3.6 shows the estimated flow at Stateline and Coulee Creek, and observed flows at Hangman Creek and Little Spokane River during 1999-2009.

3.5.3 Flow and Water Quality Data for Point Source Dischargers

There are four point source dischargers along the Spokane River that were included in the modeling efforts (see Table 3.7 for details). Each point source was characterized by flow, temperature, and additional water quality constituent concentrations.

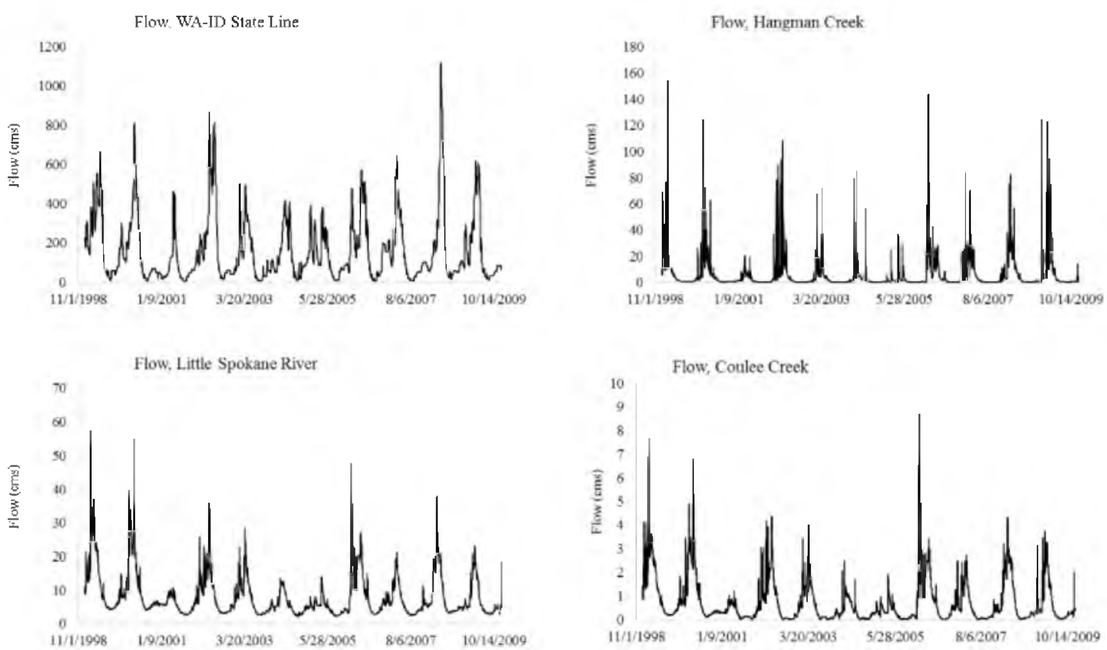


Figure 3.6 Boundary Condition Flow Data

Table 3.7 Location and Model Segment Number of Point Source Dischargers

Name of Discharger	Location	Model Segment
Liberty Lake WTP	RM 92.7	18
Kaiser Aluminum	RM 86.0	43
Inland Empire Paper Company	RM 82.6	56
Spokane WWTP	RM 67.4	115

Daily scaled flow and temperature data at these facilities were obtained from Ecology. Figure A.1 in Appendix A shows the observed flow and temperature at Liberty Lake WTP, Kaiser Aluminum, Inland Empire Paper Company, and Spokane WWTP during 1999-2009. The average discharge from these facilities was 0.035, 0.911, 0.231, and 2.001 m^3/sec, respectively. The average temperatures were 15.87, 13.64, 26.77, and 16.12 °C, respectively.

The model requires separate input files for concentrations at these facilities. The water quality data required included total dissolved solid (TDS), tracer, coliform, conductivity, chloride, inorganic suspended solid (ISS), PO_4-P, NH_4-N, NO_3-N, LDOM, RDOM, LPOM, RPOM, CBOD groups, algae groups, dissolved oxygen (DO), total inorganic carbon (TIC), and alkalinity.

Coliform data were unavailable, and thus the concentrations were assumed to be zero. Ecology used a separate CBOD compartment and CBOD ultimate data in the model to simulate organic matter originating from point dischargers. Since organic matter was accounted for in the BOD compartment, LDOM, RDOM, LPOM, and RPOM concentrations were set to zero following the Ecology model. The inorganic carbon concentrations were estimated using alkalinity data from Ecology. Inorganic suspended solids concentrations and algae concentration were set as zero. Since no data were available, tracer concentrations were also assumed to be zero. Total dissolved solid, conductivity, chloride, phosphate, nitrate, and DO were estimated by Ecology from the data provided by the dischargers. The Ecology model uses constant values for these parameters. The same was done for the extended model, keeping the values unchanged from the Ecology model. Table 3.8 shows the concentration values used for point source

Table 3.8 Concentration Values of the Point Source Discharges

Parameter	Liberty Lake WTP	Kaiser Aluminum	IEPC	Spokane WWTP
TDS	129.6	129.6	129.6	129.6
Conductivity	240.4	240.4	240.4	240.4
Coliform	0.0	0.0	0.0	0.0
Chloride	1.53	1.53	1.53	1.53
ISS	0.0	0.0	0.0	0.0
PO_4-P	0.006	0.006	0.006	0.006
NH_4-N	0.0	0.0	0.0	0.0
NO_3-N	0.895	0.895	0.895	0.895
LDOM	0.0	0.0	0.0	0.0
RDOM	0.0	0.0	0.0	0.0
CBOD	0.0	0.0	0.0	0.0
Algae	0.0	0.0	0.0	0.0
DO	8.2	8.2	8.2	8.2
TIC	18.3	18.3	18.3	18.3
Alkalinity	71.7	71.7	71.7	71.7

discharges in the extended model. All units are in mg/L except for conductivity and coliform which are in umhos/cm and MPN/100 ml units, respectively.

3.5.4 Spill and Turbine Flow

The model included four different dams in the study area – Upriver Dam (RM 80.2), Upper Falls Dam (RM 74.8), Nine Mile Dam (RM 57.8), and Long Lake Dam (RM 32.5). These dams are maintained by Avista Utilities, except for Upriver Dam which is maintained by the City of Spokane. The dams are operated as run-of-the-river facilities, so the water level at the pool behind the dam is maintained relatively constant.

Spill and turbine flow data at Upper Falls Dam, Nine Mile Dam, and Long Lake

Dam were obtained from Avista Utilities Office. The data obtained in daily total format were converted to hourly scale by dividing the daily totals by 24. For data at Upriver Dam, the City of Spokane was contacted. After discussion with the City of Spokane Office, it was found that the Upriver Dam turbine and spill flow data are approximated by interpolation from Upriver Dam Hydroelectric Project standard operating procedures and USGS maintained river flow data for the Spokane River. Although the accuracy of this method with regard to hydraulic and hydrologic dynamics is far from ideal, it is the only method currently available and will work as a good approximation of the Upriver Dam turbine and spill flows. The following method was used to obtain “turbine and spill flows”: If the river flow exceeded 7500 cfs at any given time, spill flow (cfs) = River flow – 7500 (cfs), and turbine flow (cfs) = 7500 cfs. If the river flow was equal to or less than 7500 cfs, then turbine flow = River flow (cfs), and spill flow (cfs) = 0. This was the best available method to approximate the flow dynamics at the Upriver Dam facility. The river flow data were obtained from USGS website (Station name and number, USGS 12422500 – Spokane River at Spokane, WA). Figure A.2 in Appendix A shows the observed spill and turbine flow at Upriver Dam, Upper Falls Dam, Nine Mile Dam, and Long Lake Dam during 1999-2009. The turbine flows ranged between 10 – 200 cfs, while the spill flows ranged between 0 – 1000 cfs for these dam locations.

3.5.5 Long Lake Distributed Flow and Temperature

A distributed tributary was developed for Long Lake based on flows in the Little Spokane River basin. The ratio of the drainage area surrounding the lake was divided by the drainage area of the Little Spokane River basin and then multiplied by the Little

Spokane River calculated flow. Figure A.3 in Appendix A shows the Long Lake distributed flow during 1999-2009. The distributed flow ranged from 1.7-12.6 m^3/sec (average 2.9 m^3/sec).

The Ecology model assumed the Long Lake distributed temperature to be the same as Little Spokane River temperature. Accordingly, the Little Spokane River temperature for 1999-2009, obtained from Ecology, was used as the Long Lake distributed inflow temperature.

3.5.6 Groundwater Flow, Constituent Concentration, and Temperature

The groundwater in the model was characterized for individual reaches of the river system using the model grid branches. The groundwater for each reach (branch in the model) was characterized by flow, water temperature, and water quality data. Groundwater inflows were modeled as distributed tributaries. The model did not use internal interpolation to fill in the boundary conditions between the data inputs. Details and estimates for the river inflow/outflows, assessed by Ecology staffs, are available in Annear et al. (2001).

The groundwater flow for each branch in the Ecology model is discussed in the following reports: Upper Spokane River Model: Boundary Conditions and Model Setup, 1991 and 2000; Upper Spokane River Model: Boundary Conditions and Model Setup, 2001. Not much is known about the groundwater flows for other years, particularly during high flows. As a result, the groundwater flow for other years was assumed to be the same as those calculated for 2001 in Ecology Model. More research and data collection are required to improve the groundwater representation in the model.

Groundwater constituent files for model branches between the Stateline and Nine Mile Dam were developed by Ecology from 1999 well data (also compiled by Ecology). The parameters included total dissolved solids, tracer, coliform, conductivity, chloride, ISS, phosphate-phosphorous, ammonia-nitrogen, nitrate-nitrogen, LDOM, RDOM, LPOM, RPOM, CBOD groups, algae groups, dissolved oxygen (DO), total inorganic carbon (TIC), and alkalinity.

Ecology selected water quality data from three wells in the Sullivan Road area to characterize groundwater because this was a reach where groundwater generally flows into the river (Gearhart and Buchanan, 2000). Water quality data from Ecology included total alkalinity, total dissolved solids, soluble reactive phosphorus, nitrite-nitrate, pH, temperature, chloride and dissolved oxygen.

The soluble reactive phosphorus data were used as the bioavailable phosphorus concentrations input to the model. Nitrification was assumed to be complete and ammonia-nitrogen concentrations were set to zero. Organic matter concentrations including labile and refractory dissolved organic matter, labile and refractory particulate organic matter were also assumed to be zero. Inorganic carbon (TIC) concentration was calculated from pH, alkalinity and temperature data.

In the Ecology model, groundwater constituent concentrations were held constant for the entire simulation. Consequently, the concentration of different parameters in the extended model was kept unchanged from the Ecology model and held constant over time. For groundwater temperature, Ecology used a constant 10°C for all branches. The same was done in the extended model with a time step similar to the Ecology model.

3.5.7 Meteorological Data

Meteorological data for the CE-QUAL-W2 model were obtained from the Spokane International Airport and the Spokane Felts Field. The model utilizes air and dew point temperature, wind speed and direction, and cloud cover or solar radiation. The two airports did not have solar radiation data available, so solar radiation data from Odessa, WA were used. The cloud cover data collected were not very accurate because it was measured in only a few discrete increments. The model used internal interpolation to fill in the meteorological information between input data. Meteorological data from Coeur d'Alene, ID was not used because it was too far away from the model domain, located 12 miles from the Idaho-Washington Stateline. Table 3.9 shows the required meteorological data and units for CE-QUAL-W2.

The air and dew point temperature, wind speed and direction, and cloud cover data (hourly scale) for Spokane International Airport and the Spokane Felts Field were collected from National Oceanic and Atmospheric Administration (NOAA) Data Center. Solar data (hourly scale) at Odessa, WA were obtained from AgriMet Data Center. Meteorological

Table 3.9 Meteorological Data and Units in CE-QUAL-W2

Required Data	**Units**
Air Temperature	°C
Dew point Temperature	°C
Wind Speed	m/sec
Wind Direction	radian
Cloud Cover	0 – 10 scale
Solar	Watt/m^2

data are one of the primary forcing functions of CE-QUAL-W2 (Nielsen, 2005), so it was vital that it was as frequent as possible.

The air temperature and dew point were relatively higher at Spokane Felts Field than Spokane International Airport. Wind speeds were higher at Spokane International Airport. The solar radiation at Odessa, WA ranged from 0-1059 W/m^2, with an average of 170 W/m^2. The Spokane International Airport and Felts Field use a high-speed wind gauge that only records wind speeds greater than 1.5 m/s. Wind direction was only noted for speeds greater than 1.5 m/s. The wind function requires wind speeds taken two meters from the ground (Nielsen, 2005), but the height for the wind tower used for this study is 10 meters. CE-QUAL-W2 allows the specification of the height of the wind measurements to adjust for this.

CE-QUAL-W2 requires cloud cover as a scale of 0 to 10 (0 meaning no cloud cover and 10 meaning total cloud cover). Because the cloud cover data were not recorded in this format, a conversion was required. The cloud cover data obtained from the stations were recorded as clear, scattered, broken or overcast. Table 3.10 shows the fraction of cloud cover associated with each designation for conversion to model requirements.

Table 3.10 Cloud Designations and Model Conversion Values

Type	**Fraction**	**Scale 1 -10**	**Model value**
Clear	0	0	0
Scattered	1/8 – 4/8	1.25 – 5.0	3
Broken	5/8 – 7/8	6.25 – 8.75	7
Overcast	8/8	10	10

Cloud cover in the meteorological data file was entered using the designations and the associated fractions. Because each designation spans a range of possible values, an analysis was conducted on the sensitivity of the model to cloud cover. From the results, it was found that the model is not sensitive to the change of cloud cover values within the specified ranges. Results from model runs with different cloud cover data were essentially identical. The user can specify the use of either cloud or solar data in the model. For this study, solar data were used.

Table A.1 in Appendix A complies the observed meteorological data at the two stations. Figure A.4 and Figure A.5 in Appendix A show the meteorological data at Spokane International Airport and Spokane Felts Field during 1999-2009.

3.6 Model Development for Climate Change and Population Growth Scenario

Climate change can affect the hydrological cycle by changing the runoff over watersheds and the streamflow in rivers. This can modify the pollutant's characteristics. Aggregated contribution of pollutants under low flow conditions can cause serious downstream problems regarding water quality. Therefore, climate change coupled with population growth is an important factor affecting water quantity and quality in-streams. One of the goals of this study was to assess the impacts of climate change and population changes on water quality, using the Spokane River watershed as a case study. The following section discusses the data collection and model setup for the climate change and population growth scenario simulations.

3.6.1 Climate Change Scenarios Overview

Changes in greenhouse gas (GHG, e.g., carbon dioxide, CO_2) and sulfate aerosol emissions are based on different assumptions about future population growth, socio-economic development, energy sources, and technological progress (Mote et al., 2008). Because we do not have the advantage of perfect foresight about how global, social, and economic systems will respond to emissions reduction programs, a range of assumptions about each of these factors are made to bracket the range of possible futures, i.e., scenarios (IPCC, 2013). Each of these emissions scenarios is an estimate of future emissions based on our understanding of natural sources of greenhouse gases and on assumptions about future socioeconomic trends, i.e., how much greenhouses gas will be released into the atmosphere by humans (Mote et al., 2008).

In 2010, the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES) outlined emission scenarios that were used in the IPCC's ensemble of climate models (CMIP3) for the IPCC's Fourth Assessment Report (AR4) (Mote and Salathé, 2010). SRES scenarios are grouped into scenario "families" for modeling purposes. Forty individual emissions scenarios are grouped into six families: A1F1, A1B, A1T, A2, B1, and B2. The "A" families are more economic in focus than "B" families, which are more environmentally focused. The A1 and B1 families are more global in focus compared to the more regional A2 and B2. These scenarios provide a range of changes to the climate in response to the emissions (Mote et al., 2008). More details on each scenario are present in Arnell (2004) and IPCC (2013).

Three emissions scenarios were most frequently chosen by global modeling groups in their simulations of future climate: A2, A1B (a subset of the A1), and B1. The climate forcing of all scenarios is similar until midcentury (Mote et al., 2008). Mote et al. (2008) chose A1B as the high emissions scenario and B1 as the low for analysis of 21st century Pacific Northwest climate. The Climate Impacts Group (CIG) at the University of Washington also chose A1B and B1 scenarios for its recent update of the Pacific Northwest climate change scenarios. According to the CIG, both these scenario families are considered equally probable. They analyzed simulations of future Pacific Northwest climate using 20 global climate models (GCMs) run with the A1B and B1 emissions scenarios (http://cses.washington.edu/cig/fpt/climatemodels08.shtml).

In 2013, IPCC released the Fifth Assessment Report (AR5) which was based on the fifth phase of the Coupled Model Intercomparison Project (CMIP5), which incorporates the latest versions of climate models (IPCC, 2013). A key change from CMIP3 to CMIP5 is the change in scenarios of projected greenhouse gas concentrations during the 21st century. The CMIP5 simulations are driven by representative concentration pathways (RCPs) (van Vuuren et al., 2011). The new scenarios span a range of plausible radiative forcing (Stocker et al., 2013).

RCPs take into account the impact of atmospheric concentrations of greenhouse gases and aerosols, along with the uncertainty in possible future emissions. By facilitating the coordination of new and integrated scenarios of climate, emissions, and socioeconomics, the RCPs span a wider range of possibilities than the SRES emission scenarios (IPCC, 2013). Four RCPs were developed to reflect a range of possible 21st century climate policies: RCP2.6 (mitigation scenario, very low forcing level), RCP4.5 and RCP6.0 (emissions stabilization scenarios), and RCP8.5 (very high greenhouse gas emissions) (Supharatid, 2015). By 2100, emissions peak and then decline in RCP2.6, stabilize in RCP4.5, and do not peak in RCP6.0 and RCP8.5.

Newer scenarios for very low and high greenhouse gas emissions result in a wider range among late-century warming projections for the Pacific Northwest. Unlike the SRES scenarios used in CMIP3, RCPs do not assume any particular climate policy actions. Instead, policy analysts and social scientists are free to develop mitigation scenarios that lead to one of the RCPs. The CMIP5 climate scenarios considered in this study are RCP4.5 and RCP8.5, which represent increases in radiative forcing to roughly 4.5 W/m^2 and 8.5 W/m^2 above preindustrial levels by the year 2100 (van Vuuren et al., 2011; Supharatid, 2015).

Both CMIP5 and CMIP3 datasets contain output from a large number of GCMs. CMIP5 contains more models than CMIP3 and the CMIP5 models are more advanced. Compared to CMIP3, CMIP5 models typically have finer resolution processes, incorporation of additional physics, and better-developed or well-integrated earth system components (Taylor et al., 2012; Liu et al., 2013; Miao et al., 2014). Both the CMIP5 and CMIP3 GCMs have been used to generate projections of future climate conditions across the globe. A direct comparison between the projections from the two datasets is not possible, as they use different scenarios describing the amount of greenhouse gas in the atmosphere in the future (Supharatid, 2015). Although, IPCC's Fifth Assessment Report (AR5) is based primarily on results from the CMIP5 modelling using RCPs. It also uses results from the CMIP3 modeling. The IPCC notes that, for both large scale climate patterns and the magnitudes of climate change, there is overall consistency between the projections based on CMIP3 and CMIP5 (IPCC, 2013). Both CMIP3 and CMIP5 models reproduce many characteristics of Pacific Northwest US climate fairly well including the observed Pacific Northwest seasonal cycle of wet winters and dry summers, the observed 20th century Pacific Northwest warming trend (~0.8°C/century), and observed annual temperature (Mote et al., 2013).

The CMIP5 climate scenarios based on RCP4.5 and RCP8.5 are warmer for the Pacific Northwest, on average, than the CMIP3 scenarios based on SRES-B1 and SRES-A2 (Mote et al., 2013). In the case of precipitation, both the representative CMIP3 and CMIP5 scenarios show a slightly wetter PNW future on average by mid-21st century (IPCC, 2007; Mote et al., 2013). The seasonal pattern of change in CMIP3 of slightly drier summers with slightly wetter conditions the rest of the year is also present in CMIP5 (IPCC, 2013). The SRES A1B and B1 greenhouse gas scenarios in CMIP3 are comparable to RCP6.0 and RCP4.5, respectively, in CMIP5, in terms of greenhouse gas concentrations and resultant changes in northwest climate (Maloney et al., 2014).

Efforts were made to use the latest data upon availability for this study. Meteorological data were available from CMIP5 models for RCP4.5 and RCP8.5 scenario, but flow was not. Therefore, CMIP3 data of A1B and B1 scenarios were used for projected flows. CMIP5 meteorological data for RCP4.5 scenario was used with CMIP3 flow data for B1 scenario, while CMIP5 meteorological data for RCP8.5 scenario were used with CMIP3 flow data for A1B scenario. Such a combination will do a better job of bracketing the range of plausible future greenhouse gas forcing in the Pacific Northwest. Meteorological data inputs in CE-QUAL-W2 model do not have an impact on the flow calculations; rather the flows are calculated internally using boundary condition flows (main streamflow, tributary flows, groundwater flow, and point source discharges). Meteorological data (temperature, dew point, wind speed and direction, solar radiation) drive the biological processes and rates in the model. Therefore, using meteorological data from CMIP5 and flow data from CMIP3 model will not have any disagreements in the model results. The following sections discusses more on data collection.

3.6.2 Climate Change Scenarios – Data Collection

3.6.2.1 Meteorological Data

Projected meteorological data required for CE-QUAL-W2 model included air temperature, dew point, wind speed and direction, and solar or cloud data. Projected maximum and minimum air temperature, solar radiation, and wind speed data were available in the University of Idaho (Northwest Knowledge) Multivariate Adaptive Constructed Analogs (MACA) data portal.

The Multivariate Adaptive Constructed Analogs (MACA) (Abatzoglou and Brown, 2012) method, used by the University of Idaho, is a statistical downscaling method that utilizes a training dataset (i.e., a meteorological observation dataset) to remove historical biases and match spatial patterns in climate model output. They used MACA to downscale the model output from 20 global climate models (GCMs) of the Coupled Model Inter-Comparison Project 5 (CMIP5) for the historical GCM forcings (1950-2005) and the future Representative Concentration Pathways (RCPs) RCP4.5 and RCP8.5 scenarios (2006-2100) from the native resolution of the GCMS to either 4-km or ~6-km. The MACA dataset is unique in that it downscales a large set of variables making it ideal for different kinds of modeling of future climate (i.e., hydrology, ecology, vegetation, fire, wind).

Three data products were available: MACAv1-METDATA, MACAv2-METDATA and MACAv2-LIVNEH. MACAv1-METDATA is available for the Western

USA, while MACAv2-LIVNEH and MACAv2-METDATA are available over the entire coterminous USA. MACAv2-LIVNEH and MACAv2-METDATA both use the newest version of the MACA method (version 2), while MACAv1-METDATA uses version 1. For this study, MACAv2-LIVNEH data product was used. The daily results were interpolated to hourly scale for the model input. More details on MACA downscaling, training data, MACA overview and products are available on the MACA Statistical Downscaling Method website.

The next part involved deciding which model results, out of the 20 GCMs available, to use for this study. Mote et al. (2011) provide the guideline for using climate model outputs for impact and climate diagnostics research. Selection of the model for this study was completed following these guidelines (Mote et al., 2011) and a journal article by Rupp et al. (2013). Rupp et al. (2013) evaluated the CMIP5 20th century climate simulations for the Pacific Northwest, and found out that CNRM-CM5 was generally the highest-ranked model. CESM1/CCSM4 family of models were the others that stood out as good performers. For this study, RCP4.5 and RCP8.5 scenario meteorological results from the CNRM-CM5 model have been used.

The average of maximum and minimum air temperature was used to obtain the average air temperature. Dew point temperature, required in the model, was not available. Hence, the air temperature and minimum temperature were used to calculate the dew point using the following equation (Mitchell et al., 2004; Willett et al., 2007). The minimum air temperature was considered equal to wet bulb temperature.

$$T_{Wet\ Bulb} = 0.6\ T_{Dew\ point} + 0.4\ T_{Air}$$

Projected wind direction and cloud data were not available. Observed wind direction data from 1999-2009 have been used for future scenario simulations. Sensitivity tests were performed for each meteorological variable by changing the observed data by ±10%. There was very little/insignificant change in the model results, which gave an indication that using observed data for wind direction or cloud data will not have any significant impact on model results. In another study, Perry et al. (2011) used historical wind speed and cloud cover data, due to unavailability of projected data, to simulate river water temperatures under climate change scenarios. Samal et al. (2013) also used historical regional daily meteorological data in the absence of future data. The dew point temperature for climate change simulations in studies by New et al. (2000) and Samal et al. (2013) was set equal to minimum air temperature. Table 3.11 shows the latitude-longitude of required locations for model data input.

Figure A.6 and Figure A.7 in Appendix A contain the projected meteorological data at daily scale from 2040-2050 at Spokane International Airport and Spokane Felts Field. Table A.2 in Appendix A contains the average and standard deviation of the projected meteorological data for both RCP4.5 and RCP8.5 scenarios at Spokane International Airport and Spokane Felts Field.

Projected air temperatures and dew point at Spokane Felts Field were on average higher than that at Spokane International Airport. Projected air temperature data at daily scale from 2041-2050 at WA-ID Stateline, Hangman Creek, and Little Spokane River for RCP4.5 and RCP8.5 scenarios are compiled in Figure A.8 in Appendix A. Table A.3 contains the average and standard deviation of the projected variables at different locations for both RCP4.5 and RCP8.5 scenarios.

Table 3.11 Data Station Locations

Location	Latitude	Longitude
Spokane International Airport	47.6200	117.5339
Spokane Felts Field	47.6831	117.3225
WA-ID Stateline	47.6985	117.0446
Hangman Creek	47.6546	117.4543
Little Spokane River	47.7829	117.5305

3.6.2.2 Flow Data

3.6.2.2.1 Boundary Condition

The Climate Impacts Group (CIG) predicted streamflows at 297 locations throughout the Columbia River region based on the SRES A1B and B1 global climate change scenarios. The A1B scenario typically predicts higher impacts because it assumes higher greenhouse gas emissions in the future than does the B1 scenario (Barber et al., 2011). The CIG database contains bias corrected flow estimates until year 2100 using the Variable Infiltration Capacity (VIC) hydrologic model.

Mote and Salathé (2009) ranked 20 GCMs from the IPCC AR4 based on their ability to simulate 20th century climate over the PNW. Table 3.12 shows the 10 best ranked climate models that were selected for use in the Columbia Basin Climate Change Scenarios study. These rankings are based on overall bias in the 20th century, global performance index and ability to simulate realistic 20th century North Pacific variability.

Table 3.12 Ranking of IPCC AR4 GCMs Based on Simulation Ability

Model	20^{th} C Bias	Global	North Pacific	Sum All	20^{th} C bias and North Pacific
UKMO-HadCM3	1	3	8	12	9
CNRM-CM3	2	7	4	13	6
ECHAM5/MPI-OM	3	2	3	8	6
ECHO-G	4	N/A	2	N/A	6
PCM	5	9	7	21	12
CGCM3.1	6	4	1	11	7
GCSM3	7	5	9	21	16
IPSL-CM4	8	8	10	26	18
MICROC3.2	9	6	5	20	14
UKMO-HadGEM1	10	1	6	17	16

Source: http://warm.atmos.washington.edu/2860/scenarios/

From Table 3.12, it can be seen that CGCM3.1, ECHO-G, ECHAM5, and CNRM-CM3 were found to be the top four GCMs based on their ability to simulate 20th century climate over the PNW. Based on the ranking, A1B and B1 climate scenario flow results for the CGCM3.1 model were used in this study.

Projected flows (daily scale) at WA-ID Stateline, Hangman Creek at the mouth, and Little Spokane River near Dartford were required for the CE-QUAL-W2 model. As projected flow data at WA-ID Stateline were not available, projected daily flow data at Spokane River near Post Falls and regression equation estimating flows at Harvard Road were used to determine the flows at Stateline (See Section 3.5.2.1). Projected flow data at Hangman Creek at Spokane and Little Spokane River near Dartford were used as is. The projected flow data at these locations are shown in Figure A.9 in Appendix A.

Discussion on Pacific Northwest Climate Change Scenarios on the CIG website (http://warm.atmos.washington.edu/2860/scenarios/) contains further information on the different statistical downscaling techniques used to simulate the climate data. The rationale for the selection of these downscaling approaches and their relative merits for different kinds of hydrologic analyses are discussed in the Columbia Basin Climate Change Scenarios Project report Chapter 4 (http://warm.atmos.washington.edu/2860/report/).

3.6.2.2.2 Point Source Dischargers

Projected flow data for point source discharges at Liberty Lake WWTP, Kaiser Aluminum, Inland Empire Paper Company, and Spokane River WWTP were unavailable. Among the four, the observed discharge rates for Kaiser Aluminum (0.5-1.4 m^3/sec), Inland Empire Paper Company (0.03-0.04 m^3/sec) and Inland Empire Paper Company (0.08-0.4 m^3/sec) were so small that they did not have any significant impact on the Spokane River water quality. Moreover, not much increase in land use or population is expected at these locations. Thus, observed flow data at these three locations have been used for the projected time scale.

Observed discharge rate from Spokane River WWTP varied between 1.6-2.8 m^3/sec. Increasing land use and population in the Spokane area are going to have an impact on the discharge rate from Spokane River WWTP. This scenario has been considered in the modeling.

Washington's population during 2012 was estimated at 6.8 million. According to the reports from Washington State Department of Transportation (WSDOT), this number is expected to reach 8.3 million by 2030, and 8.8 million by 2040. WSDOT projected Spokane County’s population to grow to over 500,000 by 2030, an increase of 167,576 from 2000 (40.1 % increase). According to the forecasts of Intermountain Demographics, the population in Spokane County will increase from 441,600 person in 2005 to more than 563,700 by 2030. Population forecast prepared by AVISTA Corporations place Spokane County’s population at 562,900 by 2029. Their estimate is consistent with the Washington State report (October 2007) and also historic growth patterns. State Office of Financial Management (OFM) estimates a population of 592,969 by 2040 in the Spokane County (WSOFM, 2012).

For modeling purposes, it was important to understand the growth trends in population served by the Spokane County wastewater treatment system. According to the population figures developed for the Year 2000 Comprehensive Wastewater Management Plan for Spokane County, the total population served by the Spokane County wastewater treatment system in 1999 was 53,318 (Spokane County Wastewater Facilities Plan). The population served by the WWTP is predicted to increase to 161,010 in 2020, triple the current population. The Wastewater Facilities Plan places the 2050 predicted population at 218,204, four times the current population (Spokane County Wastewater Facilities Plan). Under the future projected population scenario, wastewater flow from the Spokane Advanced Wastewater Treatment Plant (SAWTP) is expected to undergo substantial increases. Hadjikakou at al. (2011), while evaluating the impacts of future climate and environmental change on nutrient management, assumed the increases in sewage effluents to be correlated to the projections of future population increases. The average flows from the Spokane County in 2025 are projected to be 22 mgd with the combined City and County flows estimated to be 65 mgd, while the capacity of the SAWTP is currently rated at 40 mgd (Spokane County Wastewater Facilities Plan).

Discharge data at Spokane WWTP were collected from the City of Spokane to understand the change in flow rate with changing population and operation practice. When compared, it was seen that there was a clear increase in average flows from Spokane WWTP (1.4 m^3/sec in 1980s, 1.7 m^3/sec in 1990s, 2.0 m^3/sec in 2000s; an increase of appx. 20% increase in a decade). It was important to look at the projected population that will be served by the WWTP, and not just the overall population increase in the area. Subsequently, the population trend and discharge trends were used to come up with the estimates for WWTP discharges for 2041-2050 – which was 6.8-7.0 m^3/sec. The discharge estimated for 2025 using the same trends was 2.6 m^3/sec, which was close to the estimate of 2.8 m^3/sec given by the Spokane County. This provided a check for the 2041-2050 estimates used in this study.

3.6.2.3 Concentration Data

3.6.2.3.1 Boundary Condition

Concentration data at boundary condition (Washington-Idaho Stateline, Hangman Creek, Little Spokane River, and Coulee Creek) for projected time scale (2041-2050) were unavailable. To estimate the boundary condition concentrations, flow versus water quality parameter relationships were explored, similar to previous studies like Pionke et al. (1999), Chang et al. (2001), and Andrews et al. (2009). However, the correlation between the discharge and water quality parameters was not strong enough. Table 3.13 shows the correlation equations for each location, with the associated R^2 values. Flows records at Post Falls gage are primarily the same as the discharge from Lake Coeur d'Alene, which may be responsible for the weak R^2 values of the regression equations for Post Falls location. For Hangman Creek, flows during summer decrease to less than 20 cubic feet per second. As a result, there is a gradual buildup of pollutants in the watershed that do not find a way into the water bodies. Once storm events occur, part or most of these pollutants enter the system at once resulting in weak correlation between discharge and pollutant concentrations. Furthermore, the influence of groundwater on streamflows in Little Spokane River may have resulted in a weak correlation between the discharge and water quality parameters. Analysis on a seasonal basis did not provide acceptable R^2 values either. Henceforth, the weak correlation values for discharge versus water quality parameter relationships led to the decision of using the observed concentration data at boundary condition for the future timescale. Although previous studies have attempted to establish relationships between streamflow and water quality parameter loadings (Tu, 2009), it was not deemed warranted for this study.

3.6.2.3.2 Point Source Dischargers

Projected concentration data (2041-2050) at the point sources (Liberty Lake WWTP, Kaiser Aluminum, Inland Empire Paper Company, and Spokane River WWTP) were unavailable. Although projected increase in land use and population in the Spokane

Table 3.13 Correlation Equations – Discharge and Water Quality Parameters

Parameter	Post Falls		Hangman Creek		Little Spokane River	
	Equation	R^2	Equation	R^2	Equation	R^2
NH_4-N	$Y = 0.0186\ Q^{-0.062}$	0.02	$Y = 0.0163\ Q^{0.1664}$	0.14	$Y = 0.0089\ Q^{0.1689}$	0.07
NO_3-N	$Y = 0.1667\ Q^{-0.271}$	0.13	$Y = 1.0173\ Q^{0.4001}$	0.48	$Y = 1.773\ Q^{-0.269}$	0.41
PO_4-P	$Y = 0.0069\ Q^{-0.053}$	0.02	$Y = 0.0211\ Q^{0.3689}$	0.53	$Y = 0.0048\ Q^{0.4908}$	0.48
TSS	$Y = 0.9506\ Q^{0.1367}$	0.09	$Y = 5.0123\ Q^{0.6623}$	0.59	$Y = 1.9829\ Q^{0.6255}$	0.39
COND	$Y = 57.762\ Q^{-0.018}$	0.02	$Y = 294.69\ Q^{-0.199}$	0.81	$Y = 342.86\ Q^{-0.161}$	0.52
FC	$Y = 64.58\ Q^{-0.598}$	0.28	$Y = 30.475\ Q^{0.0821}$	0.01	$Y = 45.534\ Q^{-0.241}$	0.03
TURB	---	---	$Y = 3.8931\ Q^{0.8557}$	0.71	$Y = 0.4212\ Q^{0.9447}$	0.58
DO	$Y = 5.7267\ Q^{0.1273}$	0.56	$Y = 11.035\ Q^{0.019}$	0.03	$Y = 9.2185\ Q^{0.0271}$	0.04
DO *	$Y = -0.2371\ T + 13.009$	0.86	$Y = -0.1286\ T + 12.649$	0.26	$Y = 12.903\ T^{-0.129}$	0.36

Note: Y represents concentration (mg/L); Q represents flow (m^3/s); T represents temperature (°C)

area is expected to have some impacts on the Spokane WWTP discharge quality, it is very difficult to come up with estimates for the WWTP discharge concentrations. Not much increase in land use or population is expected at other three point sources. Therefore, observed concentrations from the four point sources have been used for the projected time scale. In another study by Andersen et al. (2006), point source contributions were kept constant at the base year values while assessing the climate-change impacts on hydrology and nutrients in river basin.

3.6.2.4 Stream/Water Temperature Data

3.6.2.4.1 Boundary Condition

Stream temperature data at boundary condition (WA-ID Stateline, Hangman Creek, Little Spokane River, and Coulee Creek) for projected time scale (2041-2050) was unavailable. As air temperature projections were available, they were used to estimate projected stream temperature at the required locations.

Several methods have been proposed to compute stream temperature. Stefan et al. (1980) and Stefan and Sinokrot (1993) used heat advection/dispersion transport to compute stream temperatures. Edinger et al. (1968) and Brown et al. (1971) used only surface heat transfer processes and the concept of equilibrium temperature. Other studies have used seasonal functions of stream temperature with respect to time (Hostetler, 1991; Rowe and Taylor, 1994). The simplest method for stream temperature estimation uses a linear regression between air temperatures and stream temperatures (Johnson, 1971; Stefan and Preud'homme, 1993). As estimating the effect of climate change on stream temperatures can be complex, a relationship with easily obtainable climate variables was preferred.

Linear regression models have been used to simulate stream temperatures using air temperatures (Pilgrim and Stefan, 1995; Erickson and Stefan, 1996). However, linear regression models may not accurately explain the relationship between air and stream temperature, as the relationship does not usually remain linear at the highest and lowest air temperatures (Mohsen et al., 1998). Moreover, when there are limited record lengths, the data may not level off at low or high temperatures. Although a piecewise linear regression may seem an appropriate approach to explain the nonlinear relationship between stream and air temperatures, this approach may not give representative slopes at the upper or lower ends (Mohsen et al., 1998). Mohsen et al. (1998) found that stream temperature data typically follow a general nonlinear S-shaped trend.

According to Mohsen et al. (1998), weekly stream/air temperature relationship is well described by a continuous S-shaped function. They tested several mathematical functions (Gompertz, Logistic, Richards, Morgan-Mercer-Flodin, Weibull type, etc.) to represent this relationship, and found out that logistic function parameters were the most stable and therefore selected the following from different types of logistic functions.

$$T_s = \frac{\alpha}{1 + e^{\gamma(\beta - T_a)}} \tag{3.1}$$

where T_s is the estimated stream temperature, T_a is the measured air temperature, α is the estimated maximum stream temperature, γ is a function of the slope at the point of inflection, and β represents the air temperature at the inflection point. The exponent γ is estimated as follows.

$$\gamma = \frac{4 \tan \theta}{\alpha - \mu} \tag{3.2}$$

Spokane River and its tributaries like Hangman Creek, and Little Spokane River generally do not experience freezing temperatures. To take this into account, a parameter μ was added to represent the estimated minimum stream temperature. The modified form of equation 3.1 became as follows.

$$T_S = \mu + \frac{\alpha - \mu}{1 + e^{\gamma(\beta - T_a)}} \qquad (3.3)$$

Figure 3.7 illustrates the meaning of the parameters in equation 3.3. The main advantage of this method over the linear regression is that it can better represent the tendency of some water bodies to have a threshold waters at higher air temperatures (Mohseni et al., 1999).

The CIG estimated projected changes in water temperature for various river locations throughout Washington State using air temperature data from downscaled global climate models (Mote and Salathé, 2010), based on the Mohseni approach. Water temperatures were projected using the input from 10 climate models, under two emissions scenarios (A1B and B1), for three future time periods (2020s, 2040s and 2080s). However, projected stream temperature data at locations required in this study were not present in the data base. Therefore, nonlinear regression temperature models were developed separately for WA-ID Stateline, Hangman Creek, and Little Spokane River sites following the same methodology.

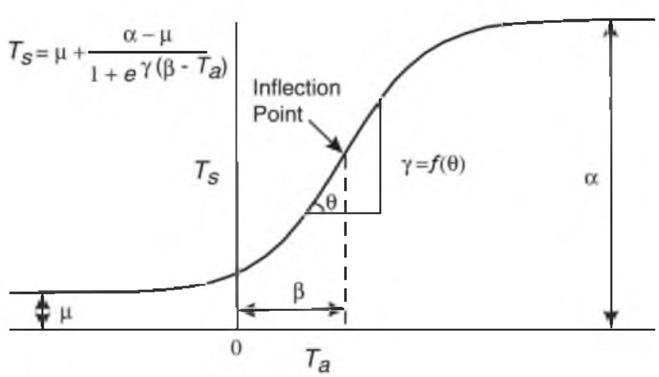


Figure 3.7 Schematic Representation of the Logistic Function Parameters

Air and stream temperature data for WA-ID Stateline (Water body ID: WA-57-1010; station location: 47.6985 latitude and 117.0446 longitude, elevation 1980 ft., RM 96.35), Little Spokane River (Water body ID: WA-55-1010; station location: 47.783 latitude and 117.531 longitude, elevation 1525 ft., RM 1.1) and Hangman Creek (Water body ID: WA-56-1010; station location: 47.655 latitude and 117.454 longitude, elevation 1720 ft., RM 0.6) were obtained from Ecology. Continuous data at 30-minute intervals for the summer months (June-September) were available for the Little Spokane River during 2002-2005 and 2007, for Hangman Creek during 2002-2007, and for WA-ID Stateline during 2001-2002. Previous studies found good correlation between stream temperatures and air temperatures at weekly timescales (Erickson and Stefan, 1996; Pilgrim and Stefan, 1995; Stefan and Preud'homme, 1993). Comparisons of daily stream temperatures to daily air temperature had higher error than those using the weekly air temperature values (Pilgrim and Stefan, 1995). According to the preference of these studies, weekly timescale was used in this study to develop the nonlinear temperature models.

The observed air and water temperatures were averaged to determine weekly temperatures for each location. The weekly average air and water temperatures were then used to develop the regression model shown in Figure 3.8. From the air temperature versus stream temperature plot, the nonlinear model parameters were estimated iteratively to minimize root mean square error (RMSE). Efficiency of fit for the nonlinear models was determined with the Nash-Sutcliffe coefficient of efficiency (NSC) (Nash and Sutcliffe, 1970). It has a maximum perfect score of 1 and no minimum, with values greater than 0 indicating satisfactory results. Root mean squared error (RMSE), mean absolute error (MAE) and mean error (ME) were also calculated following Chai and Draxler (2014).

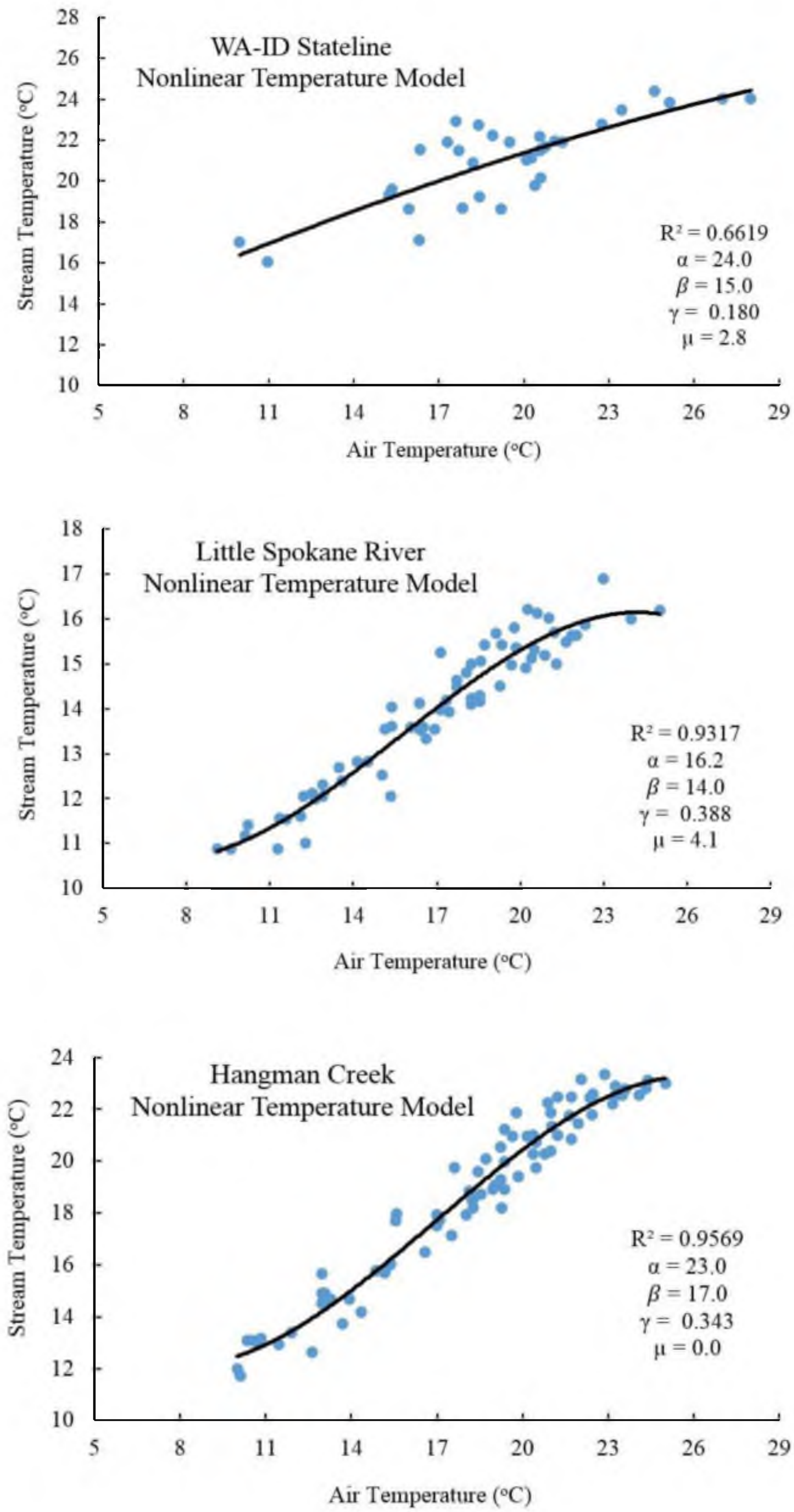


Figure 3.8 Nonlinear Regression Model Function Parameters

For Little Spokane River, the parameter values were α=16.2, β=14.0, γ=0.388, and μ=10.9 (R^2=0.93), while those for Hangman Creek were α=23.0, β=17.0, γ=0.343, and μ=12.1 (R^2=0.96). The parameter values obtained for WA-ID Stateline were α=24.0, β=15.0, γ=0.18, and μ=12.0 (R^2=0.66). The parameter values obtained for these locations were close to the results obtained for Spokane River by Mohsen et al. (1998). This was expected as both Little Spokane River and Hangman Creek are tributaries of the Spokane River. The most sensitive model parameters were the minimum and maximum stream temperature (analysis not shown).

Efficiency of fit for the models was determined with NSC, RMSE, MAE, and ME. NSE for Little Spokane River and Hangman Creek models were greater than 0.95, indicating that the model matches the data effectively. The NSE for WA-ID Stateline model was greater than greater than zero indicating a satisfactory result. Error statistics for all the models were within the acceptable limit, shown in Figure 3.9. Small differences between the RMSE and MAE indicated less variance in the individual error of the samples. Moreover, both RMSE and MAE were less than half of the standard deviation of the observed data, which is appropriate for model evaluation (Singh et al., 2004). The mean (13.95°C) and standard deviation (1.56°C) of the simulated stream temperatures at the Little Spokane River were close to the mean (13.97°C) and standard deviation (1.42°C) of the observed data. Likewise for Hangman Creek, the mean (18.76°C) and standard deviation (3.32°C) of the simulated stream temperatures were similar to the mean (18.50°C) and standard deviation (2.91°C) of the observed values. The mean (21.00°C) and standard deviation (1.27°C) of the simulated stream temperatures at WA-ID Stateline were also close to the mean (21.23°C) and standard deviation (1.66°C) of the observed data.

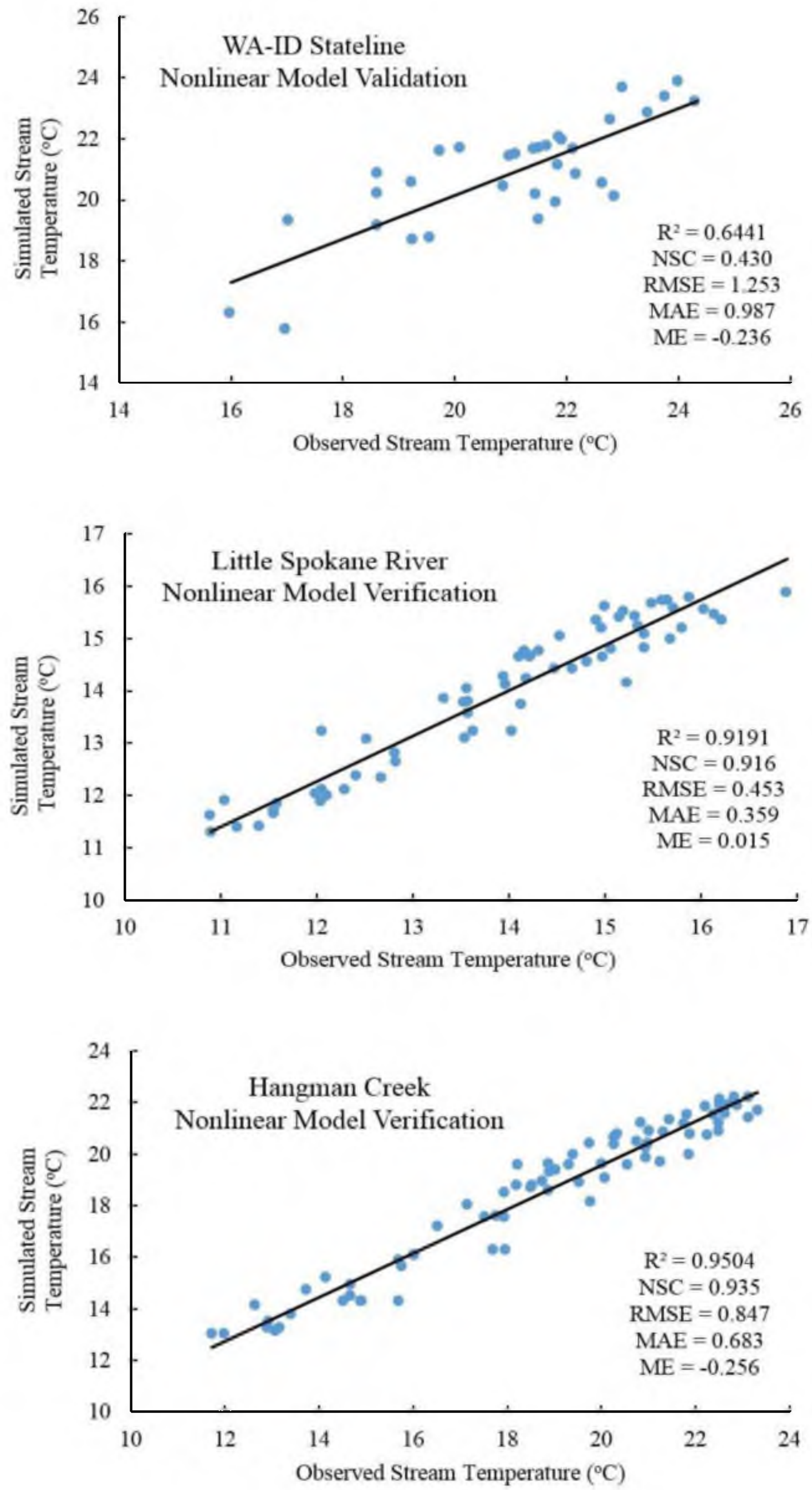


Figure 3.9 Efficiency of Fit for Nonlinear Method

Once the nonlinear models relating stream temperature to air temperature were developed for each site, they were used to estimate future stream temperatures using the projected air temperatures from CMIP5 model. Estimates of stream temperatures were obtained for both RCP4.5 and RCP8.5 scenarios. Perry et al. (2011), while simulating river water temperature under dam removal and climate change scenarios, also used the nonlinear Mohseni temperature model approach to simulate water temperatures for boundary conditions from projected air temperatures. In other studies by Sullivan et al. (2013) and Modiri-Gharehveran et al. (2014), a similar approach was adopted to estimate future steam temperatures using the estimated future air temperature. The projected water temperatures from nonlinear models are shown in Figure A.10 in Appendix A. The average and standard deviation of projected water temperatures for different locations are shown in Table A.4 in Appendix A.

3.6.2.4.2 Point Source Dischargers

Considering anthropogenic influence on point sources and their relatively small flows, water temperatures for point sources were not estimated for the future years. Historical baseline (1999-2009) values for those sources were used in the climate change scenarios. The same was done by Sullivan et al. (2013) in another CE-QUAL-W2 modeling study on the river water-quality. To check the adequacy of the assumption, the model was run by changing the discharge temperatures from different point source facilities by ±15%. This change did not influence the model outcomes, and thus contribution of the discharge temperatures was considered insignificant.

3.6.3 Climate Change Scenarios – Model Setup

Model setup for simulation of Spokane River water quality for 2041-2050 considering climate change and projected population/land use increase was completed in the same manner as for the baseline scenario. The same bathymetry, boundary condition, and initial condition used for baseline scenario (1999-2009) have been used for the climate change simulation. Changes were made in the input files (boundary condition inflow and water temperature, point source discharge, and meteorological data sets) according to the data collection or availability described in the previous sections. The calibrated model parameter values were kept unchanged. Other studies like Andersen et al. (2006) also followed this convention of keeping the model parameter estimates fixed. As projected data for the groundwater constituent concentration, temperature, and flow were not available, data for 1999-2009 have been used. The same was done for the Long Lake distributed flow and temperature.

CHAPTER 4

MODEL CALIBRATION AND SCENARIO EVALUATION RESULTS

4.1 Overview of Modeling Effort

In order to address the research objectives presented in Chapter 1, four significant modeling activities were conducted. This chapter provides explanation of:

1) Model calibration
2) Historic simulation of the 1999-2009 time period
3) Future conditions under climate change and population growth projections
4) Scenario evaluation

Prior to discussing model calibration it is important to realize that current literature lacks universally accepted procedures or guidelines for calibration (Moriasi et al., 2012). Nevertheless, there are several viewpoints among model developers and model practitioners as to how calibration should be implemented (Refsgaard, 1997; Santhi et al., 2001; Donigian et al., 2002; Jakeman et al., 2006; Moriasi et al., 2007; Jaber and Shukla, 2012; Arnold et al., 2012; Bottcher et al., 2012; Flanagan et al., 2012). Even with this large body of literature on model calibration, it is difficult to compare modeling results from different studies as users utilize different calibration methods (Moriasi et al., 2012). The approaches used to calibrate the Spokane River model in this study are explained below.

4.2 Model Calibration Procedure for CE-QUAL-W2 Study

Model calibration involves comparing observed data to modeled/predicted results until an "acceptable fit" of observed versus predicted data is achieved (Flowers et al., 2001). There are no set guidelines for determining what an adequate fit is. The user must decide if and when the model is producing useful results (Williams, 2007). For water quality investigations, Sincock et al. (2003) pointed out that hydrologic and water quality model calibration is carried out in a two-step procedure in which the flow parameters are first optimized with respect to flow, prior to the calibration of the water quality relationships. Calibration statistics mostly use the absolute mean error (AME). There are other statistics, but AME is preferred because it describes the difference between predicted and observed values (Williams, 2007; Smith et al., 2014). It is also suggested that water quality constituents be used to verify the hydrodynamic calibration.

Model calibration also requires testing model performance under conditions other than those used to generate the initial calibration model (Williams, 2007). More confidence can be placed in a model if it accurately reproduces observations from the system over a period of several consecutive years with varying hydrodynamics and water quality (Cole and Wells, 2003).

The calibration process for CE-QUAL-W2 model is presented in several studies (Berger et al., 2002; Berger et al., 2003; Annear et al., 2005; Wells et al., 2008; Williams, 2007; Hart et al., 2012; Smith et al., 2012). The general procedure of CE-QUAL-W2 calibration requires calibrating the water balance and water temperature first, and then calibrating the water-quality conditions. Since the temperature profiles change with the water quality calibration as a result of suspended solids and algae growth, the temperature calibration is typically re-evaluated after the water quality calibration (Wells et al., 2008).

Predicted water levels are compared with field data to perform water level calibration in CE-QUAL-W2. The outflows, inflows, evaporation, and residual flows are calculated and used as distributed flow for the model (Wells et al., 2008). The program 'waterbalance.exe' available with CE-QUAL-W2 can be used to estimate additional flows (either positive or negative) to match measured water levels. For water quality calibration, previous studies compared modeled vertical temperature and concentration profiles with observed data, and calculated error statistics for the profiles. They compared model time series with field data, and calculated its error. The process of model calibration described by Wells et al. (2008) is shown in Figure 4.1.

Typically, three statistics were calculated for evaluating model calibration, which include mean error (ME), absolute mean error (AME), and root mean square error (RMSE). A general rule of thumb for water quality calibration is that the absolute mean error be within 10% of the range of monitored data (Smith et al., 2012). Cumulative distribution plots can also be used.

The absolute mean error (AME) indicates the average difference between simulated and measured values. An AME of 0.5 units means that the simulated values are, on average, within ±0.5 units of the measured values (Galloway and Green, 2002). The sign of the AME indicates whether the predicted results average higher (+) or lower (-) than the observed data (Hamlin-Tillman and Haake, 1990). The absolute mean error was calculated as (Annear et al., 2005) follows.

$$AME = \frac{\sum_1^n (\text{model} - \text{observed})}{n}$$

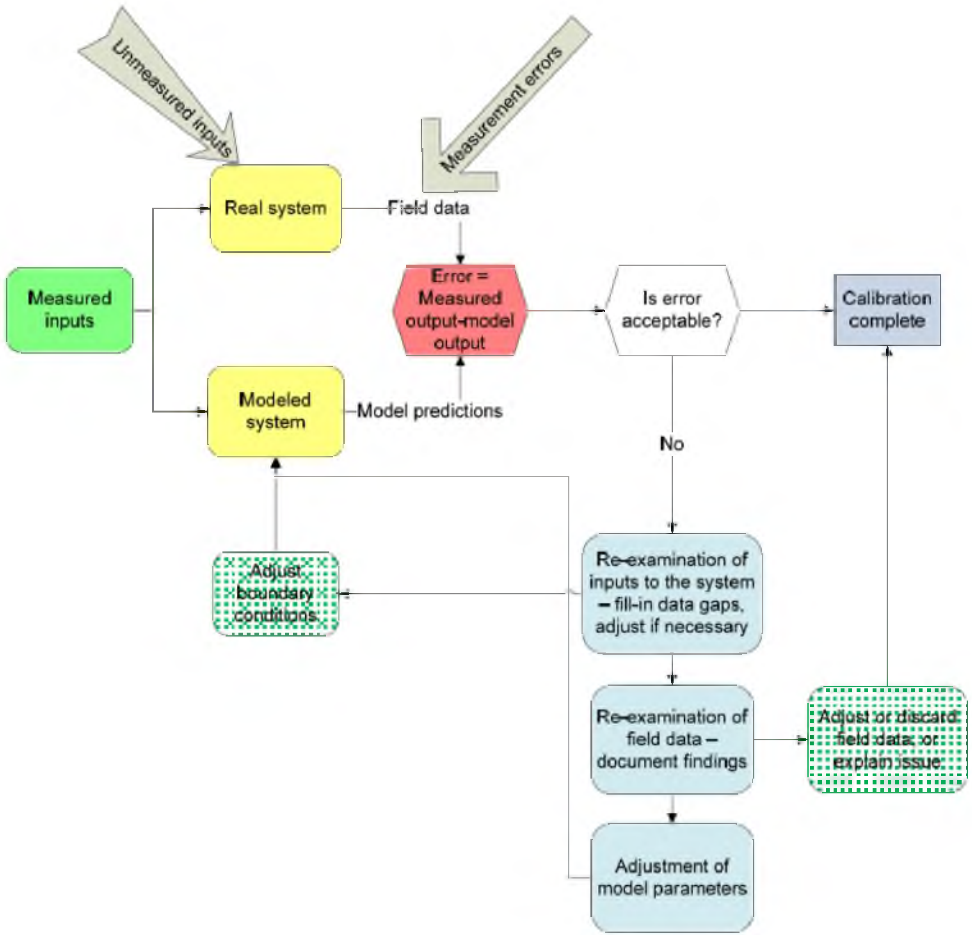


Figure 4.1 Model Calibration Philosophy (Adapted from Wells et al., 2008)

The root mean square error (RMSE) indicates the spread of how far simulated values deviate from the measured values. An RMSE of 0.5 units means that 67 percent of the simulated values are within ±0.5 units of the measured values (Galloway and Green, 2002). The root mean square error was calculated as (Annear et al., 2005) follows.

$$RMSE = \sqrt{\frac{\sum_{1}^{n}(\text{model} - \text{observed})^2}{n}}$$

AME and RMSE can be used together to analyze the variation in the errors in a set of model predictions. RMSE will always be larger or equal to the AME; the greater difference between them, the greater the variance in the individual errors in the sample. If the RMSE=AME, then all the errors are of the same magnitude.

4.3 Spokane River CE-QUAL-W2 Model Calibration

4.3.1 Historic 2001 Model Calibration

Berger et al. (2002) and Berger et al. (2003) evaluated the model calibration of the Spokane River-Long Lake system, and discussed issues relative to the calibration effort. The calibration effort was focused on model predictions of hydrodynamics (flow and water level), temperature, and eutrophication model parameters (such as nutrients, algae, dissolved oxygen, organic matter, and coliform). Berger et al. (2002) assessed the model calibration of periods from February 1, 1991 to October 31, 1991 and January 1, 2000 to October 31, 2000, while Berger et al. (2003) evaluated the model calibration of period from March 15, 2001 to October 31, 2001.

For hydrodynamic calibration, Berger et al. (2002) and Berger et al. (2003) compared the observed flow data with model results, and flow level error statistics (AME, RMSE) was calculated (Berger et al., 2002). Berger et al. (2003) reported that calibrating the flow level for extreme low flow event (such as year 2001) did not give any additional model flexibility. Nonetheless, the flow simulated in the river and its travel time were found to be were correct based on the model-data flow comparisons (Berger et al., 2003). Berger et al. (2002) and Berger et al. (2003) also discussed on the wind sheltering coefficients, groundwater inflow temperature, and reservoir outflows, essential for temperature calibration.

For the water quality calibration, the general approach was to keep the coefficient values close to commonly accepted literature values. If during the calibration process, a particular combination of coefficient values did not produce good results, values were changed back to their default values, and a new avenue was investigated for better model outcome (Berger et al., 2002).

For various water quality parameters, time series and vertical profile data were compared with model results, and AME and RMS error statistics were calculated. Berger et al. (2002) also computed sensitivity coefficients to determine sensitivity of the predictions of dissolved oxygen, temperature, chlorophyll a, and periphyton biomass to model parameter values. Groundwater concentration data were sparse; thus, they were adjusted to match the observed values. This is an important limitation in the prediction of impacts of future conditions on river water quality. Parameters that were important in the model calibration efforts of Berger et al. (2002) and Berger et al. (2003) included dissolved oxygen reaeration equations, periphyton growth rates, periphyton half saturation parameters for phosphorus and nitrogen, ammonia-nitrogen preference equation for periphyton, and the stoichiometry of the periphyton.

The key objective of Ecology was to have the CE-QUAL-W2 model represent a good approximation of the major forcing processes and features of the system that affect water quality particularly related to the primary and secondary processes that control dissolved oxygen in the Spokane River and Long Lake. Consequently, the best available information ranging from historical information, collected field data, and laboratory data to the selection of literature values to define specific model parameters, rates, and constants were used in calibrating the CE-QUAL-W2 model (Cusimano, 2004).

4.3.2 Spokane River Extended CE-QUAL-W2 Model Calibration

Extending the model beyond the 2001 water year simulation required examining the calibration parameters based on data collected during other time periods. This section evaluates the model calibration and discusses issues relative to that calibration effort. The calibration effort was focused on model predictions of hydrodynamics, temperature, and eutrophication model parameters (such as nitrate, ammonia, phosphate, and dissolved oxygen). The model calibration period was from March 15, 1999 to December 31, 2009. The monitoring sites utilized in the model calibration consisted of sites along the Spokane River, tributaries and point discharges to the river. Berger et al. (2002) and Berger et al. (2003) contain further description of these monitoring sites. The following sections contain details on the hydrodynamic, temperature, and water quality calibration for this study.

4.3.2.1 Hydrodynamic Calibration

Flow data available at Upper Falls Reservoir (segment 86, RM 74.8), Spokane River at Spokane (USGS Station 12422500, segment 97, RM 72.9), Spokane River at Barker Rd (USGS Station 12420500, segment 24, RM 90.3), Nine Mile Reservoir (segment 151, RM 57.8), and Long Lake (segment 188, RM 32.2) were used for the hydrodynamic calibration. The flow error statistics calculated for these locations are tabulated in Table 4.1. The absolute mean error (AME) range from 4.5–21.9 m^3/s from upstream to downstream locations. The modeled flows were on average within 10% of the observed data, which fulfilled the requirement provided by Smith et al. (2012). Figure 4.2 shows the time series flow calibration plot for Spokane River at Spokane site. Flow calibration plots for rest of the location are available in Appendix B (Figure B.1), which also shows near perfect match of the model results and observed data. Consequently, the flow calibration was deemed satisfactory.

Table 4.1 Flow Error Statistics for the Spokane River Sites, 1999-2009

Location	AME (m^3/s)	Average % Error
Spokane River at Barker Rd	4.5	5.6
Spokane River at Spokane	9.2	4.7
Upper Falls Reservoir	10.7	4.8
Nine Mile Reservoir	15.4	7.6
Long Lake	21.9	7.5

Number of Data Compared: 3954

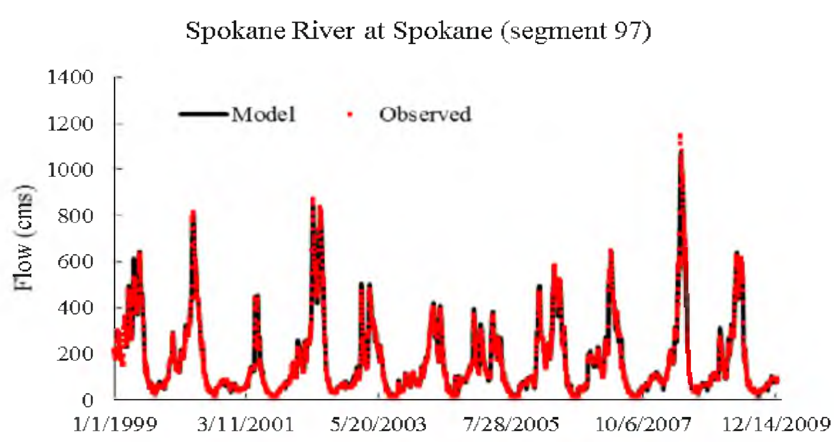


Figure 4.2 Flow Prediction Compared with Data for the Spokane River at Spokane

4.3.2.2 Temperature Calibration

Berger et al. (2003) found Spokane River temperature calibration to be sensitive to wind sheltering coefficients, groundwater inflow temperature, and the accurate representation of reservoir outflows. Values of temperature calibration parameters from the Ecology model were kept unchanged in this study. Berger et al. (2003) discussed in detail how these parameter values were obtained or estimated. Temperature calibration values from the Ecology model were kept unchanged for this study.

4.3.2.2.1 Vertical Profiles

Model output temperature profiles from different sampling site locations were compared with observed data. Table B.1 in Appendix B lists the sites at Spokane River-Long Lake where temperature profiles were collected. Figure 4.3 shows vertical temperature profiles from 2000 and 2001 at Long Lake-Station 3 (Segment 168). The rest of the vertical profiles are available in Appendix B (Figure B.2 to Figure B.16). Table B.2 in Appendix B shows the overall error statistics for all sites. Temperature profile AME statistics were in the range 0.36-1.72°C, which is close to those previously obtained for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003).

Water temperatures were generally higher near the surface, and lower towards the bottom layers. With increasing depth, less radiation reached the bottom layers making the water cooler than the surface. The result of downward diffusion of heat and upward vertical advection of colder deeper water created the vertical temperature profile shape (Talley et al., 2011). The shape of vertical temperature profiles matched closely to the expected pattern.

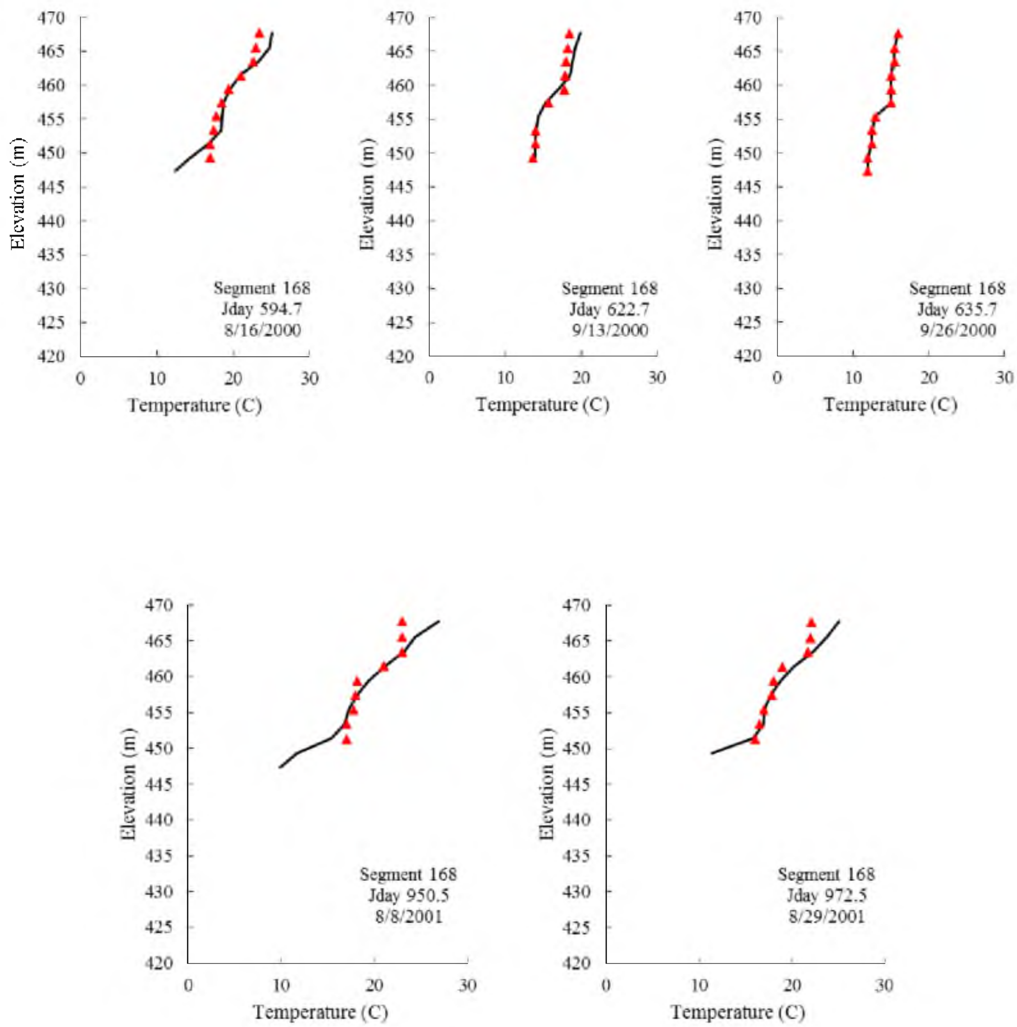


Figure 4.3 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and Observed Data (Red Dots) at Long Lake at Station 3 (Segment 168)

4.3.2.2.2 Time Series

Stream water temperature fluctuates between day and night (diurnal temperature changes) and over longer time periods (seasonally). In the spring, snowmelt running into rivers reduced the water temperature to below the ambient air temperature. Water temperatures increased during summer due to increase in solar radiation and decrease in streamflow.

Table B.3 in Appendix B shows the list of sites where water quality time series data (temperature, dissolved oxygen, nitrate-nitrogen, ammonia-nitrogen, and phosphate-phosphorous) for Spokane River were collected during 1999-2009. Figure 4.4 compares time series temperature model results with observed data at Riverside State Park site (segment 119). The rest of the time series temperature data comparison and time series temperature error statistics for all sites is available in Appendix B, Figure B.17 and Table B.4, respectively.

Temperature time series absolute mean error statistics were in the range 0.4-1.7^{o}C, which is analogous to those obtained by Berger et al. (2002) and Berger et al. (2003) for the Spokane River-Long Lake. Moreover, the comparison of mean and standard deviation of the model calculated and observed time series temperature data (Table B.5, Appendix B) pointed to resemblance between the data sets, indicating that the model was well calibrated.

The model successfully reproduced the expected pattern in water temperatures over the annual cycle. Characteristically, the surface temperature profiles exhibited the warmest temperatures during months of July and August with the coolest temperatures in January. Summer-time river temperatures routinely exceeded the state water quality standard of

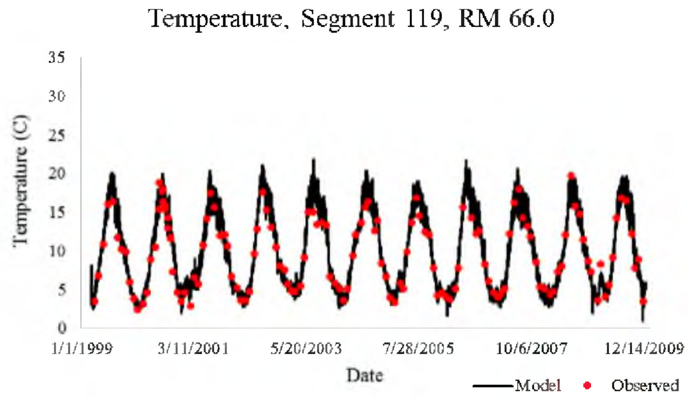


Figure 4.4 Time Series Comparisons of Temperature Data at Riverside State Park Site

20°C (Cusimano, 2003) at upstream stations such as Baker Road and Sullivan Road. Water temperature decreased after Sullivan Road till Nine Mile (due to groundwater inflow), and then started to increase with proximity to the Long Lake. Downstream sites also indicated occasional exceedance of the water quality criteria set for temperature in the Long Lake reservoir and Spokane River. The similarity of upstream and downstream temperatures indicated short retention time (Reference: Bryson Project).

4.3.2.3 Water Quality Calibration

The general approach toward water quality calibration was to keep coefficient values close to commonly accepted literature values (Berger et al. 2003). Some of the important water quality parameters values used during the calibration by Berger et al. (2002) and Berger et al. (2003) are shown in Table 4.2. The complete list of model calibrated values can be found in Cole and Wells (2000), Berger et al. (2002) and Berger et al. (2003).

Table 4.2 Spokane River CE-QUAL-W2 Model Water Quality Parameters

Variable	Description	Units	Typical values*	Calibration Values
AT11	Lower temperature for algal growth for algal type 1	ºC	10	8
AT21	Lower temperature for maximum algal growth for algal type 1	ºC	30	10
AT31	Upper temperature for maximum algal growth for algal type 1	ºC	35	20
AT41	Upper temperature for algal growth for algal type 1	ºC	40	30
AK11	Fraction of algal growth rate at ALGT1 for algal type 1	-	0.1	0.1
AK41	Fraction of algal growth rate at ALGT4 for algal type 1	-	0.1	0.1
AK21	Fraction of maximum algal growth rate at ALGT2 for algal type 1	-	0.99	0.99
AK31	Fraction of maximum algal growth rate at ALGT3 for algal type 1	-	0.99	0.99
ET11	Lower temperature for Periphyton growth for Periphyton type 1	ºC	10	1
ET21	Lower temperature for maximum Periphyton growth for Periphyton type 1	ºC	30	3
ET31	Upper temperature for maximum Periphyton growth for Periphyton type 1	ºC	35	20
ET41	Upper temperature for Periphyton growth for Periphyton type 1	ºC	40	30
EK11	Fraction of Periphyton growth rate at ALGT1 for Periphyton type 1	-	0.1	0.1
EK21/ EK31	Fraction of maximum Periphyton growth rate at ALGT2/ ALGT3 for Periphyton type 1	-	0.99	0.99
EK41	Fraction of Periphyton growth rate at ALGT4 for Periphyton type 1	-	0.1	0.1

Table 4.2 Continued

Variable	Description	Units	Typical values*	Calibration Values
SDK	Sediment decay rate	/day	0.06	0.10
PARTP	Phosphorous partitioning coefficient for suspended solids	-	1.2	0
NH4DK	Ammonia decay rate (nitrification rate)	/day	0.12	0.40

4.3.2.3.1 Dissolved Oxygen Calibration

Previous applications of CE-QUAL-W2 have shown DO to be a better indicator of proper hydrodynamic calibration than temperature (Cole and Wells, 2000), as it is much more dynamic than temperature (Flowers et al., 2001). Dissolved oxygen can be sensitive to inflow boundary conditions and algae and organic matter internal processes, including sediment oxygen demand (Wells et al., 2008).

4.3.2.3.1.1 Vertical profiles. Vertical DO profiles are mainly controlled by physical factors such as surface reaeration and river flow. Vertical mixing of DO from surface reaeration and photosynthesis sets a higher DO concentration near the surface and lower near the bottom (Sarmiento and Gruber, 2005; Lin et al., 2006). Photosynthesis produces oxygen and remineralization consumes it. These processes were reflected in the vertical profiles of oxygen, with lower concentrations at depth than at the surface. Sometimes, strong river flow can cause lower DO concentrations near the surface and higher near the bottom (Lin et al., 2006).

Figure 4.5 shows the dissolved oxygen profile data comparison with model results at Long Lake-Station 2 (Segment 174) (See Appendix B, Figure B.18 to Figure B.32 for rest of the vertical DO profile calibration plots). Table B.6 in Appendix B shows the AME and RMS error statistics for the dissolved oxygen vertical profiles. Dissolved oxygen AME statistics were in the range 0.02-1.51 mg/L, which is comparable to the errors obtained for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003).

Vertical profiles show that the dissolved oxygen (DO) tends to violate the minimum criterion (8.0 mg/L) at the river sites (from Idaho border to Nine Mile Bridge) with proximity to the Long Lake.

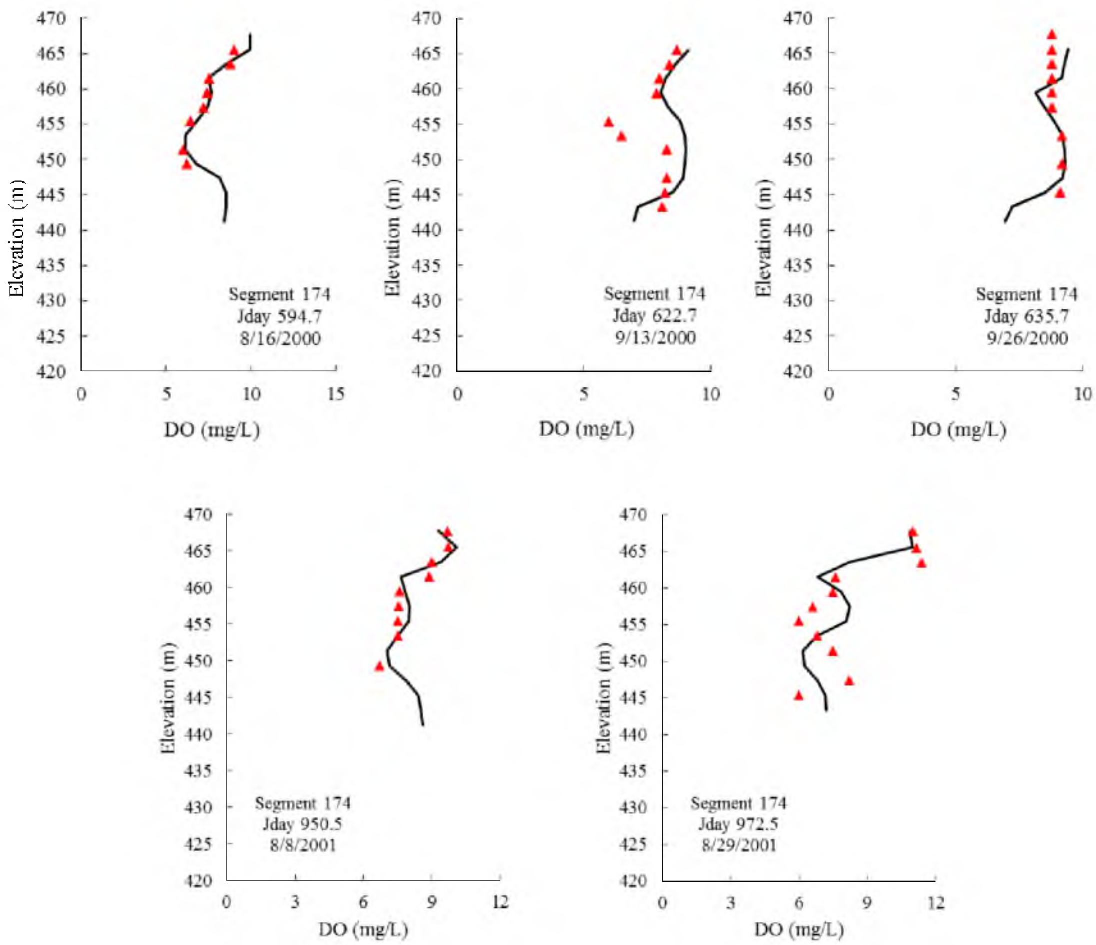


Figure 4.5 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 2 (Segment 174)

<u>4.3.2.3.1.2 Time series</u>. The amount of oxygen that dissolves in water varies in daily and seasonal patterns, and decreases with higher temperature, salinity, and elevation (Carr and Neary, 2008). Dissolved oxygen can be reduced to very low levels during the winter months when water is trapped under ice.

Figure 4.6 compares time series dissolved oxygen model results with data at Riverside State Park site (Segment 119) (See Appendix B, Figure B.33 for the rest of the time series DO profile calibration plots). Table B.7 in Appendix B shows the time series DO error statistics for all data sites in 1999-2009. Time series AME statistics were in the range 0.21-1.05 mg/L, which is comparable to the calibration errors previously obtained for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003). Close match between the mean and standard deviation of the model calculated and observed DO data (Table B.8, Appendix B) indicated that the model was well calibrated for dissolved oxygen.

A cyclic pattern was observed in the surface dissolved oxygen time series, where minimum dissolved oxygen concentrations in river typically was observed during late summer. During spring, dissolved oxygen levels declined rapidly and remained low during summer. At warmer water temperatures, oxygen holding capacity of water decreased (Solheim et al., 2010; Chang et al., 2015), and biological metabolism of algal and microbial community increased (Arvola et al., 2010; Moore and Ross, 2010). In addition, deposition of organic matter, coupled with limited reaeration during summer (reference: Maryland DNR) caused the dissolved oxygen concentrations to be lowest during August and September.

In colder fall months, a rapid return to the higher oxygen levels was observed.

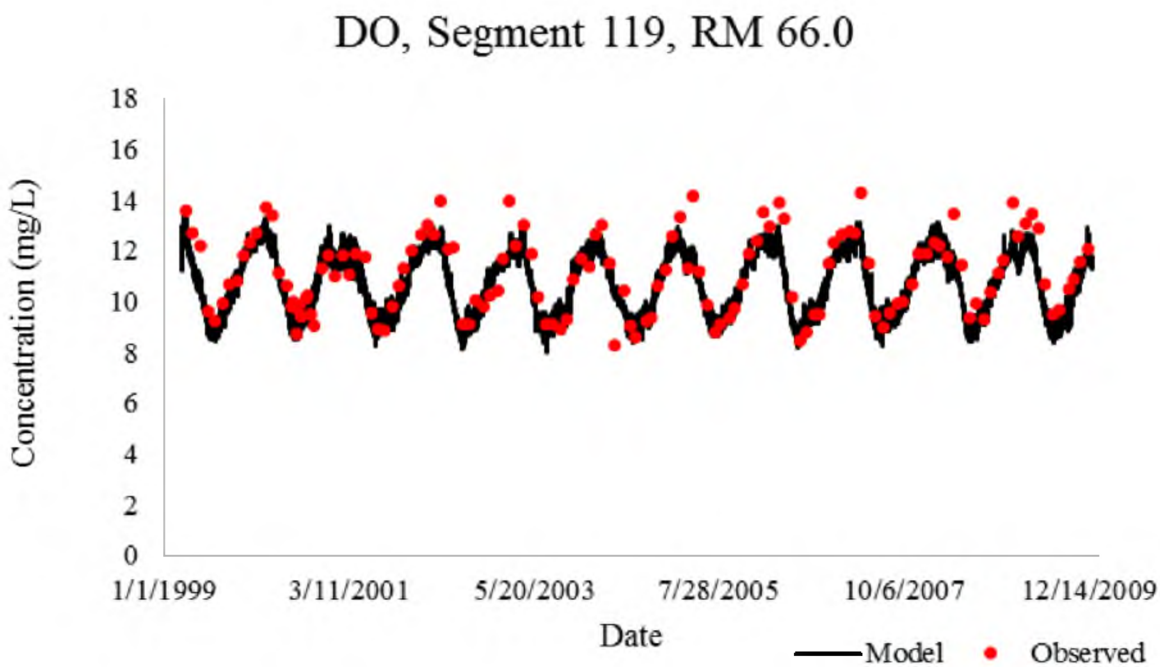


Figure 4.6 DO Time Series Comparisons Data at Riverside State Park Site

Enhanced reaeration and the physical ability of water to hold more oxygen at colder temperatures (Solheim et al., 2010) contributed to this seasonal patterns.

Dissolved oxygen (DO) time series revealed that the minimum criterion of 8.0 mg/L for the river (from Nine Mile Bridge to the Idaho border) was routinely exceeded at most of the sites during summer. Dissolved oxygen dropped around Upriver Dam and Green Street Bridge stations, and started to rise afterwards. After crossing the Spokane WWTP, dissolved oxygen decreased continuously until Long Lake.

4.3.2.3.2 Nitrate-Nitrogen Calibration

4.3.2.3.2.1 Vertical profiles. The concentrations of nitrate-nitrogen in the lower water column are generally controlled by a combination of incoming nutrient levels and the reflux of surface waters (Khangaonkar et al., 2012). Due to well mixed conditions, reflux of surface waters into lower layers can reduce the bottom concentrations (Khangaonkar et al., 2012).

Figure 4.7 compares the nitrate-nitrogen profile data and model results at Long Lake-Station 1 (Segment 180) (See Appendix B, Figure B.34 to Figure B.38 for rest of the vertical nitrate-nitrogen profile calibration plots). Table B.9 in Appendix B shows AME and RMS error statistics for the nitrate-nitrogen vertical profiles. The AME compares well with the previously obtained error statistics for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003).

Due to high levels of algal growth associated with spring and summer blooms (Moore and Ross, 2010; Arvola et al., 2010), the concentrations of nitrate near the surface layers were depleted as seen from the vertical profiles. Toward the bottom layers, condition became anoxic/anaerobic, and denitrification occurred (Seitzinger et al., 2006; Ghane et al., 2015). This may have reduced nitrate concentration in bottom layers for some instances (Segment 174, 180, 187).

4.3.2.3.2.2 Time series. Figure 4.8 compares time series nitrate-nitrogen model results with data at Sandifer Bridge site (Segment 97) (See Appendix B, Figure B.39 for the rest of the nitrate-nitrogen time series calibration plots). Table B.10 in Appendix B shows time series nitrate-nitrogen error statistics for all sites in 1999-2009. Nitrate-nitrogen time series AME statistics were in the range 0.038-0.533 mg/L, which is similar to the

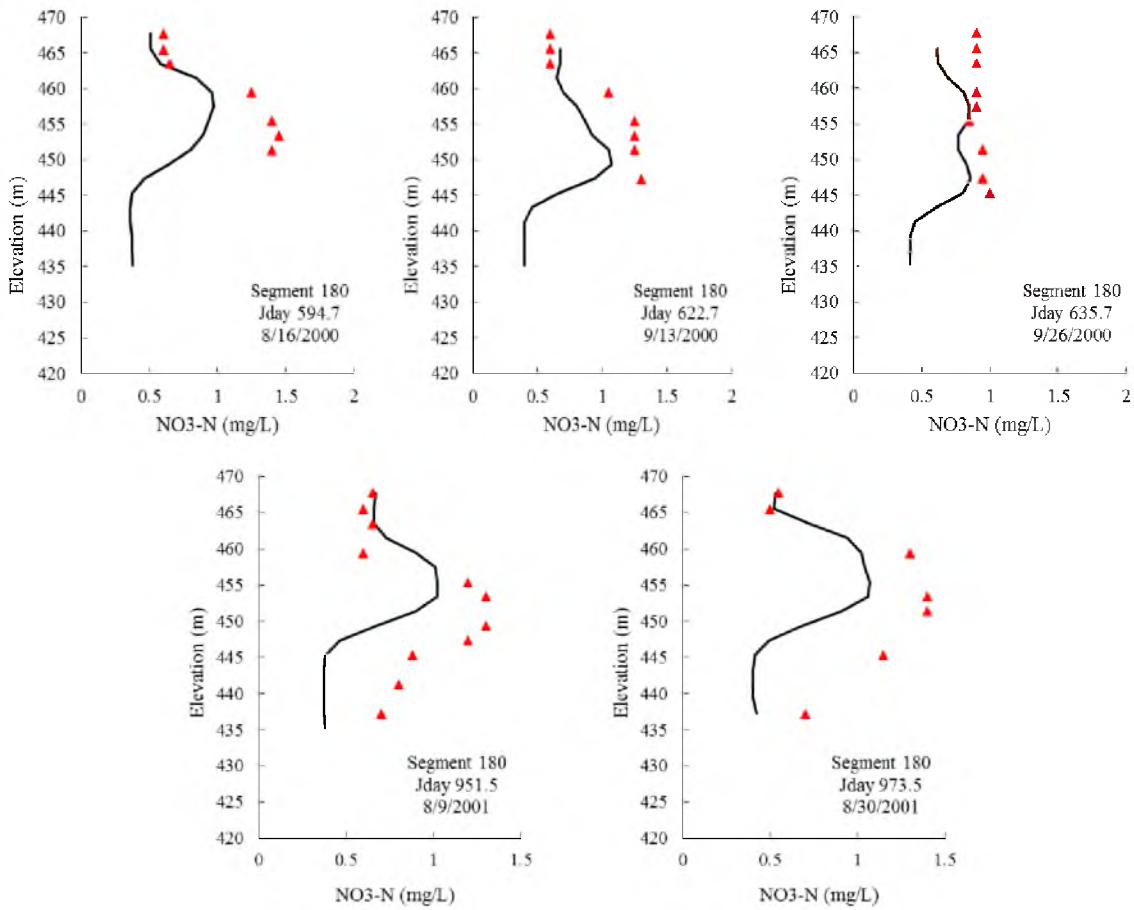


Figure 4.7 Comparison of Model Predicted Vertical Nitrite-Nitrate Nitrogen Profiles (Black Line) and 2000 and 2001 Data (Red Dots), Long Lake at Station 1 (Seg 180)

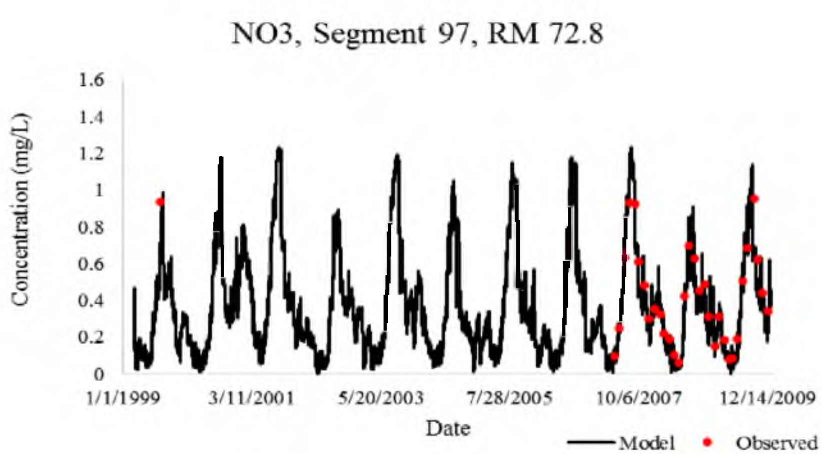


Figure 4.8 Nitrate-Nitrogen Time Series Comparisons at Sandifer Bridge Site

calibration errors obtained for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003). Moreover, the likeness between the mean and standard deviation of the model calculated and observed nitrate-nitrogen (Table B.11, Appendix B) indicated acceptable model calibration of nitrate-nitrogen.

Nitrate concentrations at the surface were the greatest during the winter, potentially because of a reduction in denitrification rates and decrease in algal uptake during the winter (Bark, 2010). Concentration decreased during spring at most sites, which coincided with increased streamflow during the snowmelt runoff or spring storms indicating dilution (Bark, 2010). Nitrate concentration continued to decrease during summer. This happened potentially because of the reduction in nitrate inputs (from decreased surface runoff) and increase in biological uptake (Fenelon, 1998; Lee et al., 2012). In terms of the location, nitrate-nitrogen concentrations increased from upstream to downstream stations, with a small drop at the Spokane WWTP location.

4.3.2.3.3 Ammonia-Nitrogen Calibration

4.3.2.3.3.1 Vertical profiles. Ammonia-nitrogen vertical profiles were collected in Long Lake during some days 2000 and 2001. Figure 4.9 compares the ammonia-nitrogen profile data and model results at Long Lake-Station 0 (Segment 187) (See Appendix B, Figure B.40 to Figure B.43 for rest of the nitrate-nitrogen vertical profile calibration plots). Table B.12 in Appendix B shows AME and RMS error statistics for the ammonia-nitrogen vertical profiles. Ammonia-nitrogen profile AME statistics were in the range 0.008-0.033 mg/L, which is like those obtained for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003). The shapes of the vertical ammonia-nitrogen concentration

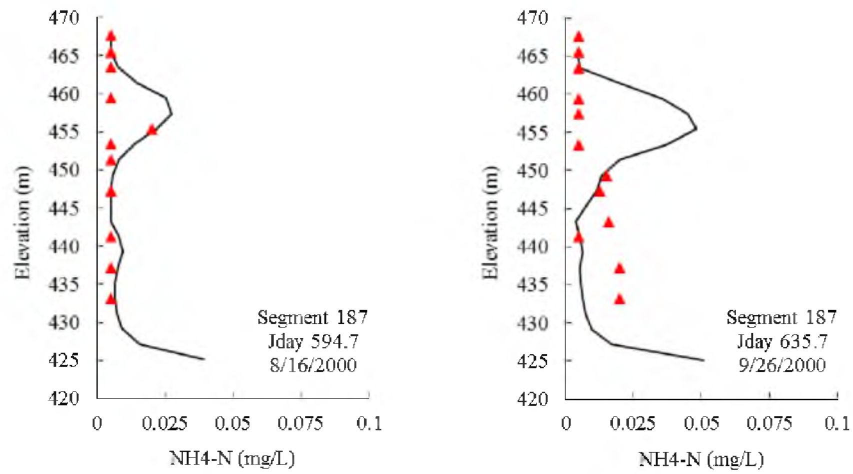


Figure 4.9 Comparison of Model Predicted Vertical Ammonia Nitrogen Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 0 (Segment 187)

profiles were similar to those of nitrate-nitrogen.

Algae spores (first stage in algae life cycle) consume ammonia-nitrogen for growth (aquarium-fertilizer.com), and their effect on ammonia concentrations in the surface layers, where algal blooms occur (Khangaonkar et al., 2012), can be noticed from the vertical profiles. Organic matter in streamflow and a significant portion of the phytoplankton biomass upon decay settled at the bottom, where they were remineralized to inorganic nutrients (ammonification). Oxygen depleted conditions in bottom layers (Segment 174, Segment 180, Segment 180) may have favored ammonification, increasing the ammonia-nitrogen concentration in the bottom layers (Lillebø et al., 2007). On a seasonal basis, ammonification rates were generally highest in summer and lowest in spring or fall (Lillebø et al., 2007). The alternating curved shapes may also have been formed due to the influence of organic and inorganic forms of nitrogen coming from tributaries and point sources, and the thermal stratification affecting biological processes (Pastuszak, 1995).

4.3.2.3.3.2 Time series. Figure 4.10 compares the ammonia-nitrogen time series data and model results at Long Lake Dam (Segment 188) (See Appendix B, Figure B.44 for rest of the nitrate-nitrogen time series calibration plots).

Time series AME statistics (Table B.13, Appendix B) for ammonia-nitrogen (0.002-0.023 mg/L) compares well with the AME obtained for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003). Moreover, similarity between the mean and standard deviation of observed and model calculated value (Table B.14, Appendix B) shows that the model was well calibrated for ammonia-nitrogen.

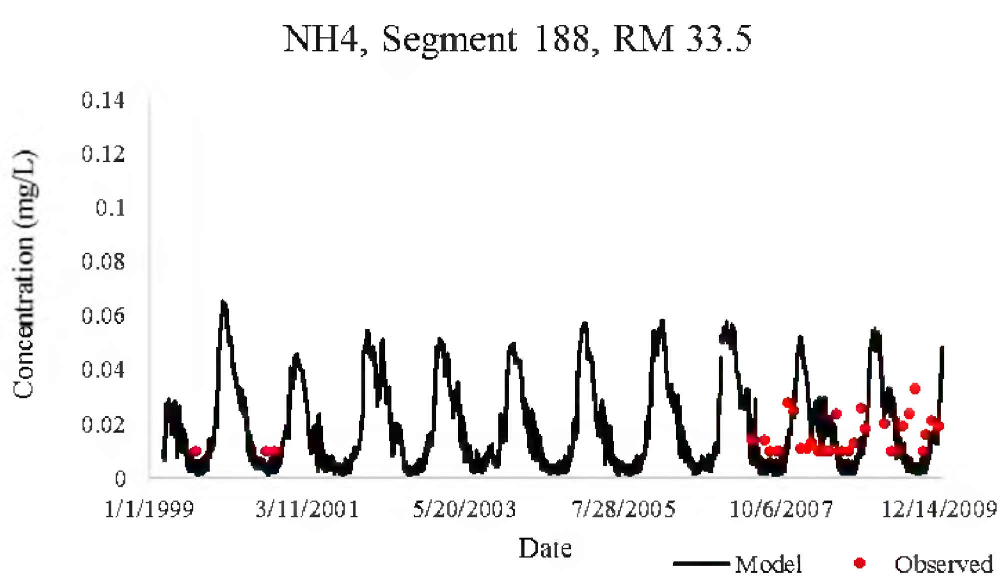


Figure 4.10 Ammonia-Nitrogen Time Series Comparisons at Long Lake Dam

Ammonia concentrations at surface peaked in winter and decreased rapidly by spring or early summer. Higher concentrations in winter may primarily have been due to its accumulation in stream water in the winter under ice and snow cover because of limited algal metabolism and increased mineralization of decaying organic matter under reducing conditions within stream bottom sediments (Lee et al, 2012). Increase in algal activity during the growing season (spring and early summer) (Arvola et al., 2010) resulted in decreased ammonia concentration. Concentrations decreased in spring also because of the dilution effect from spring runoff events (Lee et al, 2012).

4.3.2.3.4 Phosphate-Phosphorous Calibration

4.3.2.3.4.1 Vertical profiles. Soluble reactive phosphorus (SRP) vertical profiles were collected in Long Lake in 2000 and 2001 for some days. No additional vertical profiles were collected upstream of Long Lake. Figure 4.11 show SRP vertical profile data and model results at Long Lake-Station 1 (Segment 180) (rest profiles in Appendix B, Figure B.45 to Figure B.49). Table B.15 in Appendix B shows AME and RMS error statistics for the SRP vertical profiles. Phosphate vertical profiles absolute mean error statistics were in the range 0.002-0.007 mg/L, which is comparable to those previously obtained for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003).

The vertical profiles showed evidence of summertime phosphate consumption in the surface layers. Algal blooms are generally known to occur in the top 5 – 20 m (Khangaonkar et al., 2012), and their effect on nutrient concentrations such as phosphate consumption for growth was easily noticeable. The stations where phosphate was depleted

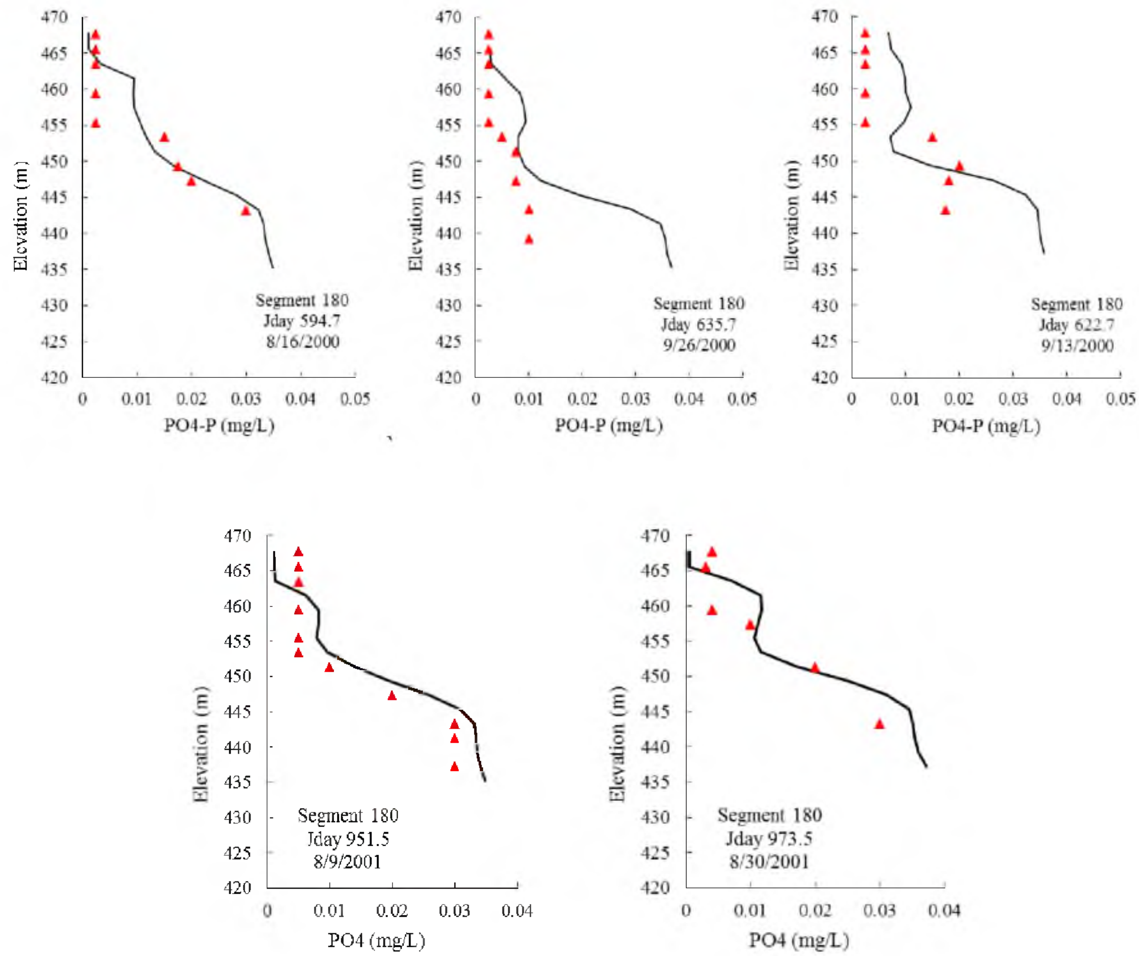


Figure 4.11 Comparison of Model Predicted Vertical Phosphorus Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 1 (Seg 180)

were the same locations where nitrate were reduced. River runoff, sediment resuspension and sedimentation could also have had important impacts on sediment behavior and regulated phosphate distributions and shaped their vertical profiles (Khangaonkar et al., 2012). This, along with complex mixing, makes phosphate kinetics in the deeper layers difficult to interpret. Dissolved inorganic phosphate resuspension fluxes tend to be highest at locations experiencing summer anoxia (reference: Maryland DNR). This may have been the reason for higher phosphate concentrations in the bottom layers (Segment 168, 174, 180, and 187). The phosphate concentrations in the euphotic zone of the reservoir seemed to have exceeded the criterion set for the period of June 1 to October 31.

4.3.2.3.4.2 Time series. Figure 4.12 compares phosphate-phosphorous time series model results with the observed data at Long Lake Dam site (Segment 188). The rest of the time series plots are placed in Appendix B (Figure B.50). The phosphate-phosphorous time series absolute mean error statistics (0.001 - 0.015 mg/L), shown in Table B.16 (Appendix B), compare well with the calibration errors obtained previously for the Spokane River and Long Lake by Berger et al. (2002) and Berger et al. (2003). Furthermore, similarity between the mean and standard deviation of observed and model calculated values (Table B.17, Appendix B) shows that the model was well calibrated for phosphate-phosphorous.

Phosphorus concentration patterns and the type of phosphorus present changes with changing hydrologic conditions and seasons (Lee et al., 2012). Phosphate concentrations tended to be greater in late spring to summer. Concentrations may have increased in the stream during the late summer when surface runoff contributed less of the total streamflow, and groundwater containing phosphate became a more dominant source in streams during

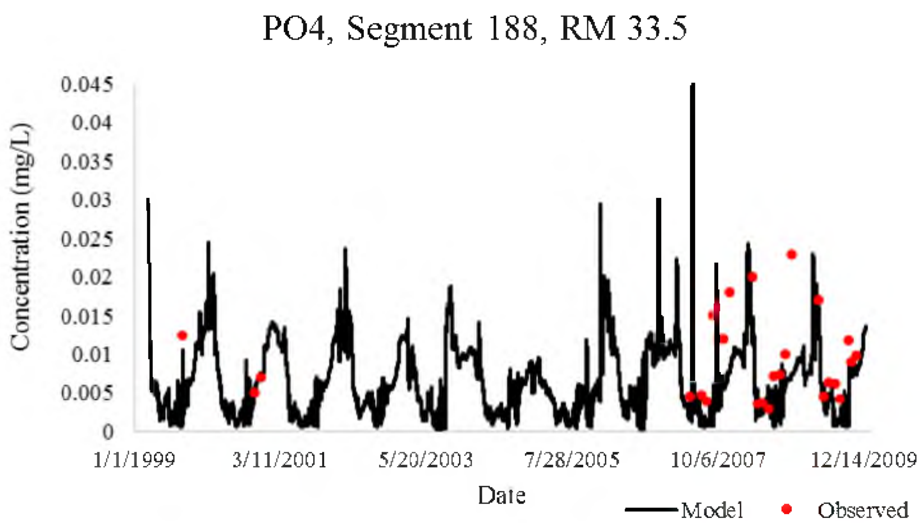


Figure 4.12 Phosphate Time Series Comparisons at Long Lake Dam

low flows (Lee et al., 2012). In terms of the location, phosphate concentrations kept increasing from upstream to downstream stations, with a small drop at Spokane WWTP location. The water quality criteria for phosphate concentration were routinely exceeded at the reservoir sites.

4.3.2.4 Overall Calibration

Table 4.3 shows a summary of the model errors for the parameters of interest in the extended Spokane River-Long Lake model from this study. The minor differences between RMSE and AME indicate small variances in the individual errors in the sample. Both RMSE and AME for parameters of interest in the extended model were very close to the previous model calibration errors in 2000 (Berger et al. 2002), 2001 (Berger et al. 2003)

Table 4.3 Model Errors in the Spokane River-Long Lake System

Parameter	Overall RMSE	Typical range RMSE	Overall AME	Typical range AME
Temperature, ºC	1.25	0.14 – 3.01	0.96	0.14 – 1.75
Dissolved oxygen, mg/L	0.86	0.03 – 1.76	0.64	0.02 – 1.51
Nitrate-Nitrogen, mg/L	0.24	0.05 – 0.56	0.21	0.04 – 0.53
Ammonia-Nitrogen, mg/L	0.021	0.002 – 0.043	0.015	0.002 – 0.033
Phosphate, mg/L	0.006	0.002 – 0.022	0.005	0.001 – 0.015

and 2001/2004 (Annear et al., 2005) Spokane River models, and furthermore met the criteria set by Wells et al. (2008).

In general, the model reproduced well the river and reservoir responses to the known boundary. The calibration errors from this study were also comparable to model calibration errors in other similar modeling studies (Hanna and Campbell, 2000; Wells et al., 2000; Berger et al., 2001; Berger and Wells, 2005; Sullivan and Rounds, 2005; Annear et al., 2006), shown in Table B.18 (Appendix B). This gives an impression that the extended Spokane River-Long Lake model has been well calibrated, and is well suited for evaluating the impacts of management strategies to improve water quality in the Spokane River-Long Lake region.

4.4 CE-QUAL-W2 Modeling Results for 1999-2009 Time Period

Spokane River management decisions have traditionally been based on short term simulation. The issue with short term simulation is that it is a portion of the real picture displaying results from only a small time period, in this case a low flow period. But the nutrient dynamics may not be sensitive to the short time spans, particularly PO_4-P. Inflow PO_4-P has a tendency to remain imbedded in the sediments, which may be resuspended in the following seasonal cycle depending on the flow and oxygen levels. Therefore, with short term simulation of water quality lies the possibility of misinterpreting the impacts of nutrient loading from previous years. Moreover, nonpoint nutrient loads are typically lowest in low flow years, and therefore are not entirely appropriate for long term management decisions encompassing varying flow states. Loading reductions, although they could produce preferable results for a low flow year, may not be adequate for high flow years carrying higher nutrient loads.

Results from the calibrated Spokane River model for 1999-2009 were examined with respect to Washington State water quality standards, particularly DO, temperature, and PO_4-P; and several instances were found where predicted temperature and concentrations were in violation of the standards. While the TMDL year (2001) represented a low flow year; model simulated temperatures, dissolved oxygen and nutrient concentrations in other years were equally bad or worse than 2001.

According to the model results in Figure 4.13, DO standard was never violated during 2001 at Long Lake. But long term simulation results revealed instances of DO standard (8 mg/L) violations during September of 1999, 2002, 2003, 2006, and 2008; and October of 2005 and 2009. The reason for the DO standard violation during late summer

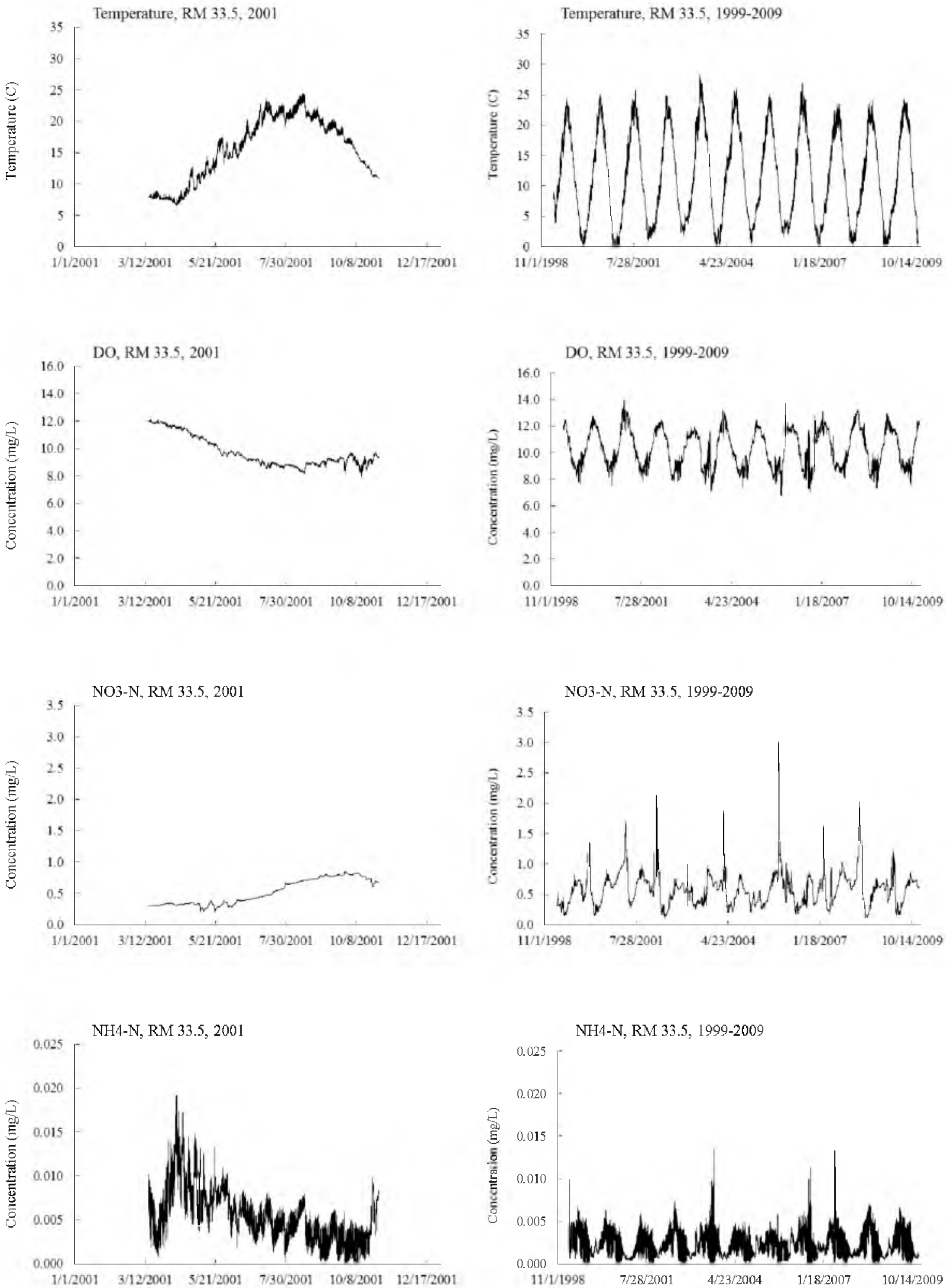


Figure 4.13 Comparing Long Term Simulation to Short Term Simulation

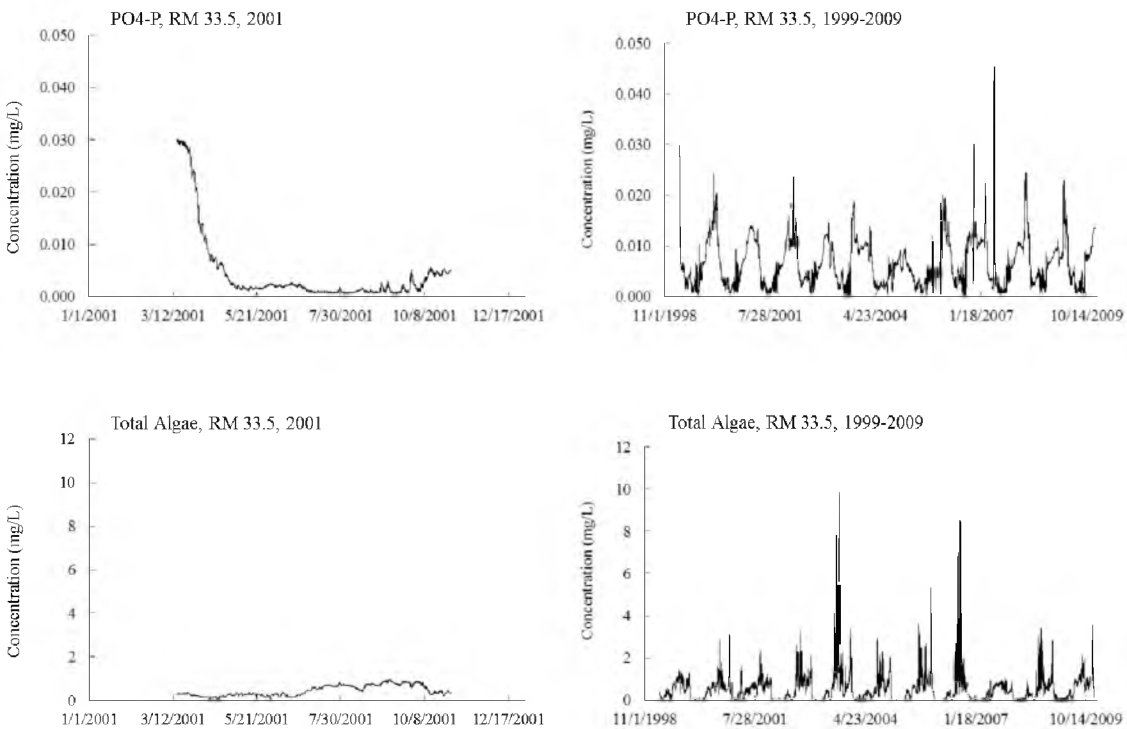


Figure 4.13 Continued

is the reduced streamflows, coupled with higher temperatures and greater microbial activity consuming the oxygen. More importantly, the seasonal swing of dissolved oxygen ranged from 7-14 mg/L in the long term simulation, while that for the short term simulation was only between 8-12 mg/L. Therefore, the short term simulation failed to capture the extreme drops in DO due to the presence of higher algae in the system.

For phosphorous, the only instance of standard violation at the Long Lake during 2001 was the initial period of the model run, occurring due to the model's initial condition (Figure 4.13). The concentrations during rest of 2001 was well below the 0.025 mg/L criterion. However, long term simulation indicated several high phosphate concentration peaks over the 1999-2009 period, with violations of the standard during February of 2000, March of 2002, November of 2006, May of 2007, and March of 2008. Furthermore, several nitrate-nitrogen peaks were observed in the long term simulation during February to April, all of which were considerably higher than 2001 concentrations (Figure 4.13). This implies that higher winter flows were accompanied by higher phosphorous and nitrogen loads from nonpoint sources (Chang et al., 2001; Andrews et al., 2009; Bark, 2010; Lee et al., 2012), a fact that did not become apparent from the short term simulation. Contribution ratios of nutrient loads from point sources increased as streamflow decreased, while contribution ratios from nonpoint loading increased as streamflow increased (Du et al., 2014). The availability of higher nutrients during the high flow years, coupled with high temperatures, resulted in significant algal blooms at the Long Lake, which the short term simulation was unable to capture (Figure 4.13).

With 90% and 50% load reductions from point and nonpoint sources, respectively, the state water quality standards were easily met for low flow year, 2001. The relatively small nonpoint loading contribution from low flows is apparent from Figure 4.14. However, the loading reduction did not solve all the problems over a long term period 1999-2009. Dissolved oxygen levels regularly violated the standard, and instances of phosphate standard violation were still observed. During calibration, it became apparent that the model is fairly sensitive to nonpoint loadings, for which data are sparse. The model results with loading reduction scenario are shown in Figure 4.14.

The calibrated CE-QUAL-W2 model setup of the Spokane River for the observed 1999-2009 period was used as the "baseline" scenario for comparison with model simulations of climate change and population growth scenarios.

4.5 Predicting Climate Change and Growth Impacts

4.5.1 Base Case Model Simulation

The calibrated Spokane River CE-QUAL-W2 model was used to simulate projected river water quality during a 2040-2050 time frame under future climate change scenarios. Model simulations were completed for both high and low emission scenarios. The details of these emission scenarios have been discussed previously in Chapter 3. Spokane River at Spokane (RM 72.0) and Long Lake (RM 32.5) sites were selected as the river and reservoir locations for comparison of the climate induced results. Several simulations were completed, including the "base case" and alternative scenario evaluation. The "base case" simulation is for the climate change and population growth scenarios for 2040-2050, which should not be confused with the "baseline" scenario model simulation for 1999-2009 with observed values.

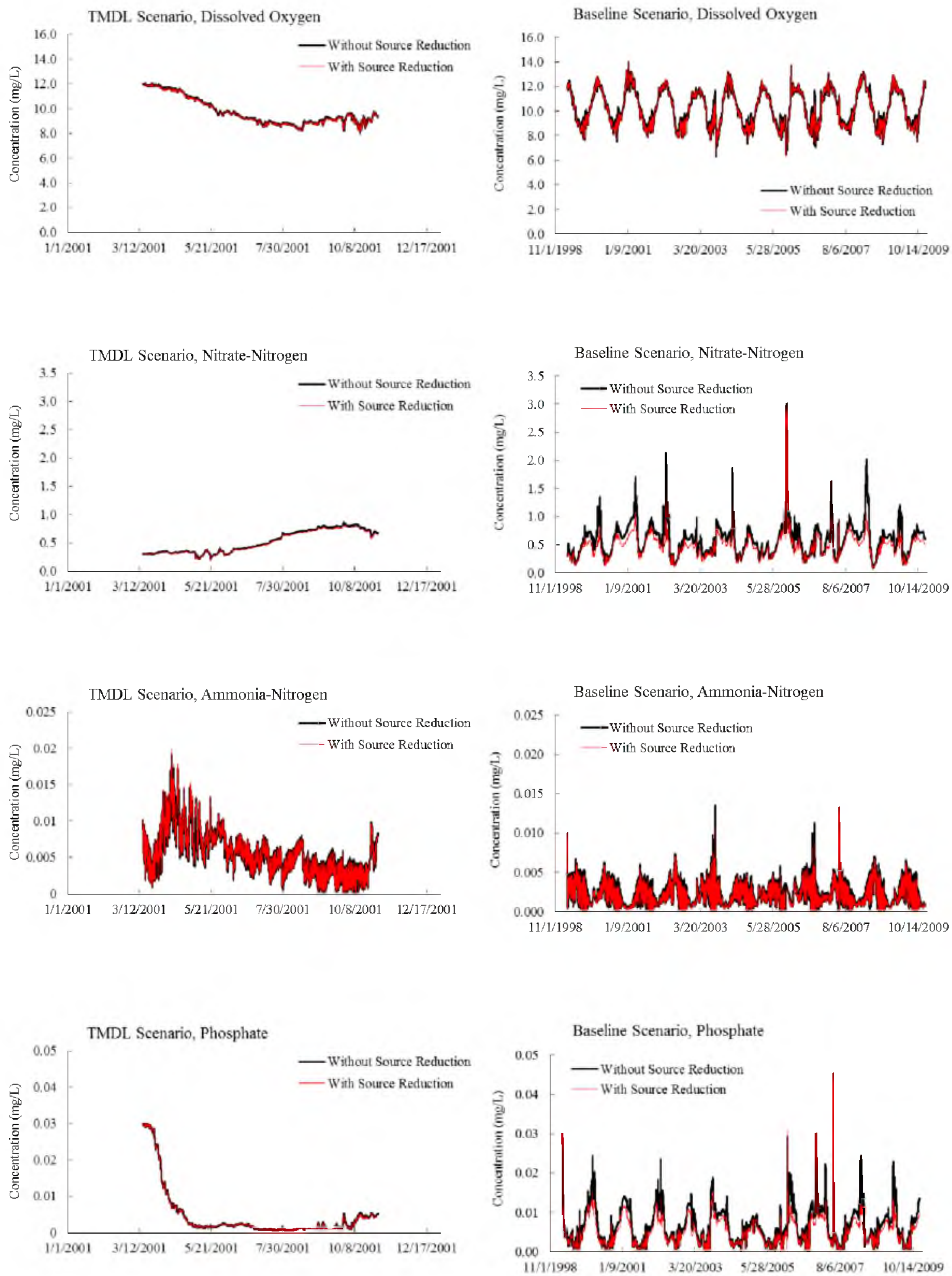


Figure 4.14 Impact of Point and Nonpoint Source Loading Reduction, Long Lake Site

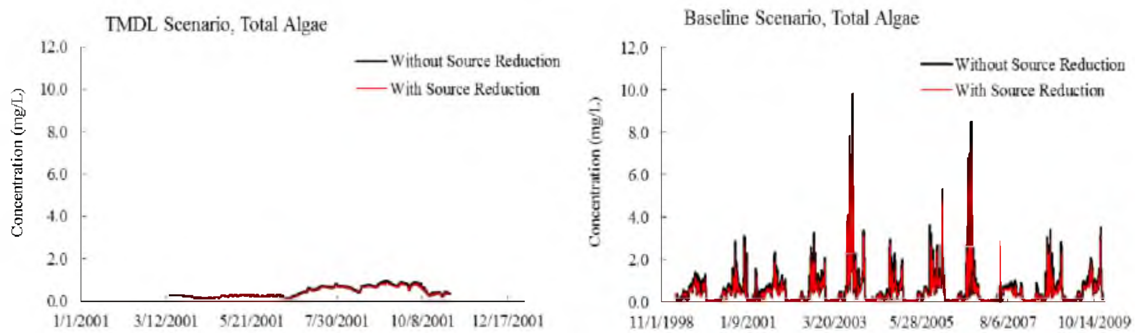


Figure 4.14 Continued

4.5.1.1 Flow

The Climate Impacts Group worked with several prominent water management agencies in the Pacific Northwest to develop hydrologic climate change scenarios for different streamflow locations in the Columbia River basin. Streamflow projections for the Spokane River from their work were used as flow input in this study. Simulation of modified climate (using flow inputs from CMIP3 CGCM3.1) predicted an increase of streamflow in Spokane River-Long Lake during 2040-2050 for both high and low emission scenarios. The increase in rainfall for both these scenarios (Mote et al., 2013) most likely resulted in an increase in proportion of the water lost through surface runoff in the CMIP3 CGCM3.1 model. In addition to increased precipitation, the increase in streamflow may also be associated with surface geology (Chang et al., 2001). Fu (2005) and Fu et al. (2007) previously predicted increases in annual streamflow in the Spokane River Watershed for 2020s and 2040s due to the projected increase in precipitation and temperature. Much of the projected increase in streamflow, as found from this study, is expected to occur during the winter months, while flows during the summer months are likely to remain similar to the baseline scenario. This implies that the issue with low summer flows in the Spokane River still persists under the modified climate. The relatively higher increase in winter streamflows in the Spokane River was also projected by Fu (2005).

A two-tailed paired t-test was conducted, where the streamflow differences between the baseline and the climate scenarios at different locations along the Spokane River were found to be statistically significant (0.05 significance level, $p < 0.001$). Figure 4.15 shows the projected streamflows at Spokane River at Spokane and Long Lake. Although the streamflows increased on average in the Spokane River-Long Lake system, the increase

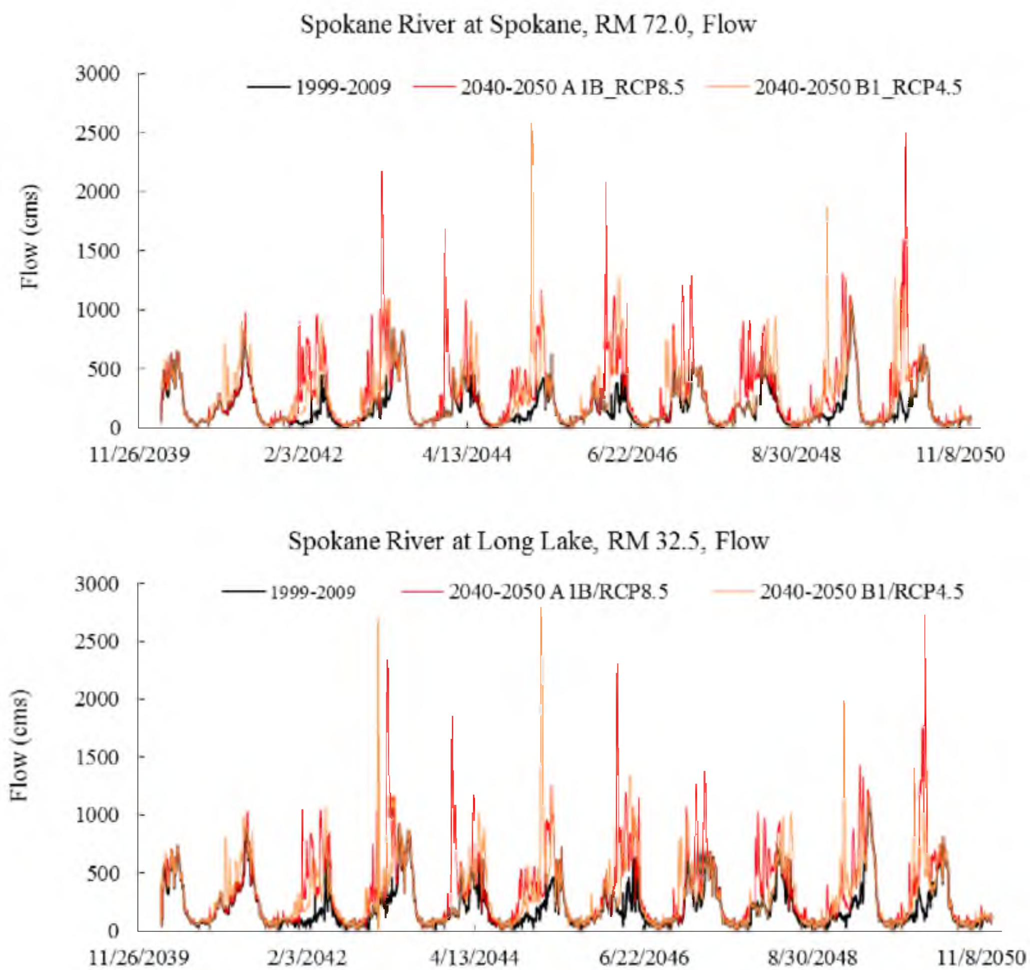


Figure 4.15 Climate Change Model Simulation Results for Flow

was not proportionate throughout the year. The projected monthly flows at Spokane River at Spokane and Long Lake site, shown in Figure 4.16, point toward the disproportionate increase in streamflows during December–April. Such disproportionate increase in streamflows may cause exceedingly high nutrient loading during winter/spring (Andrews et al., 2009; Bark, 2010; Lee et al., 2012), which may overwhelm the existing pollution control infrastructure. Moreover, occurrence of higher nutrient loading during winter may also have consequences on water quality during summer.

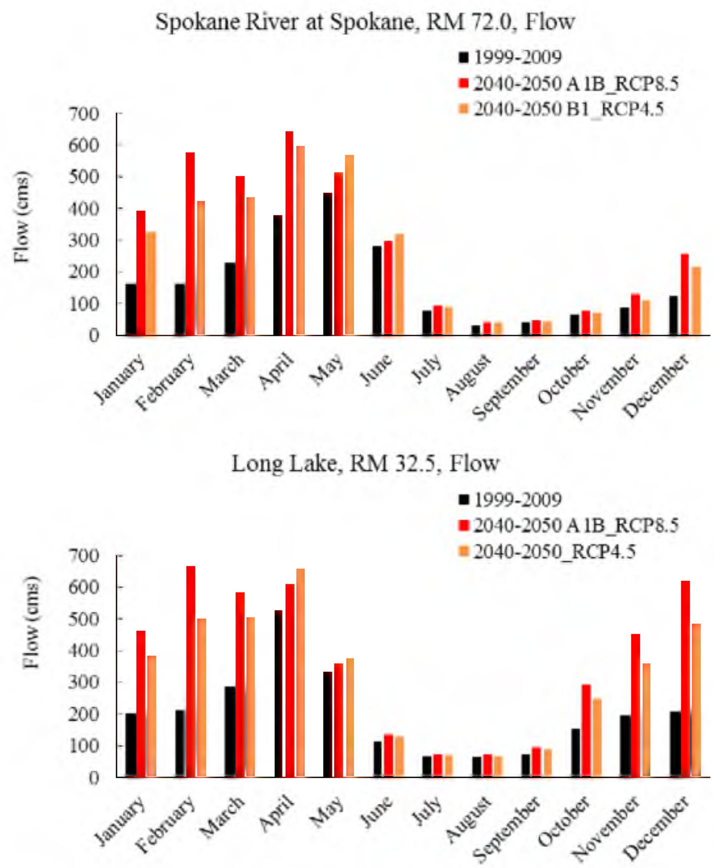
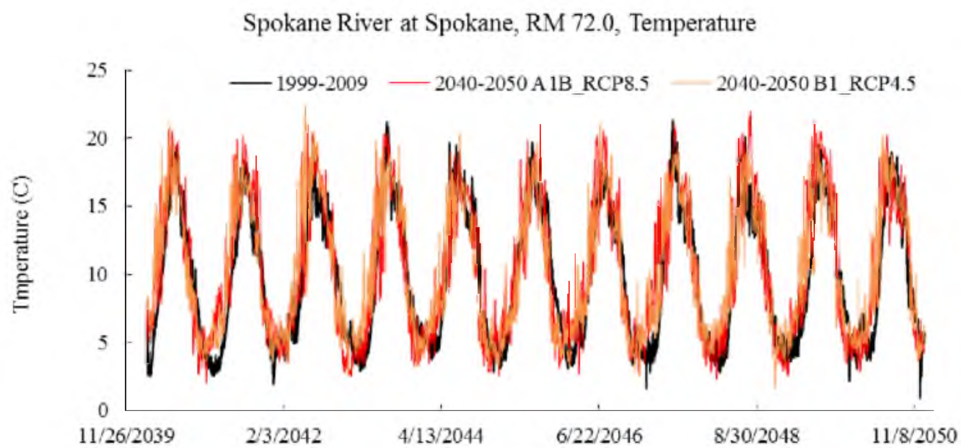


Figure 4.16 Comparison of Flow Results at Monthly Scale

4.5.1.2 Stream Temperature

Water temperature at surface in the Spokane River and Long Lake, as seen from the climate change scenario simulations, is expected to increase during 2040-2050 due to the projected increase in air temperature. Figure 4.17 shows the projected stream temperatures at Spokane River at Spokane and Long Lake. Temperature projections showed occasional violation of the temperature criteria (20°C) at river sites during summer months, but Long Lake temperatures regularly exceeded the criteria during summer for both high and low emission scenarios.



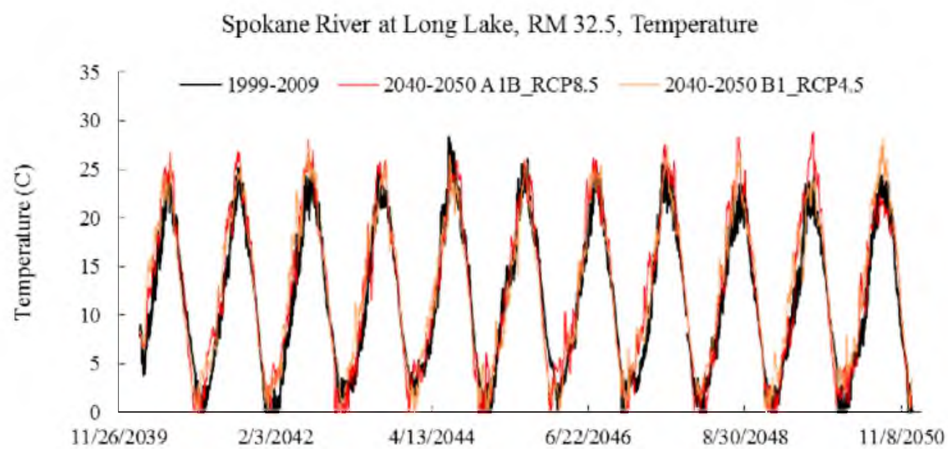


Figure 4.17 Climate Change Model Simulation Results for Temperature

Compared to the baseline scenario, water temperature at river locations is expected to increase by about 0.6–0.8ºC for the high and low emission scenarios during 2040-2050; while at Long Lake, the increase is about 1.8–2.2ºC. A decreasing trend was seen in the temperatures from Stateline to Nine Mile due to groundwater influence. A two-tailed paired t-test supported the fact that the temperature differences between the baseline and climate scenarios at river locations were statistically significant (0.05 significance level, $p < 0.01$).

Figure 4.18 compares the projected monthly water temperatures at Spokane River at Spokane and Long Lake, which indicated that temperature increased in general for all months. Violation of the temperature criteria under climate change impacts was also apparent from the vertical plots at Long Lake, shown in Figure 4.19. Compared to the baseline scenario, the projected temperatures were higher at deeper layers at Long Lake, suggesting the possibility of lower dissolved oxygen and higher algae growth.

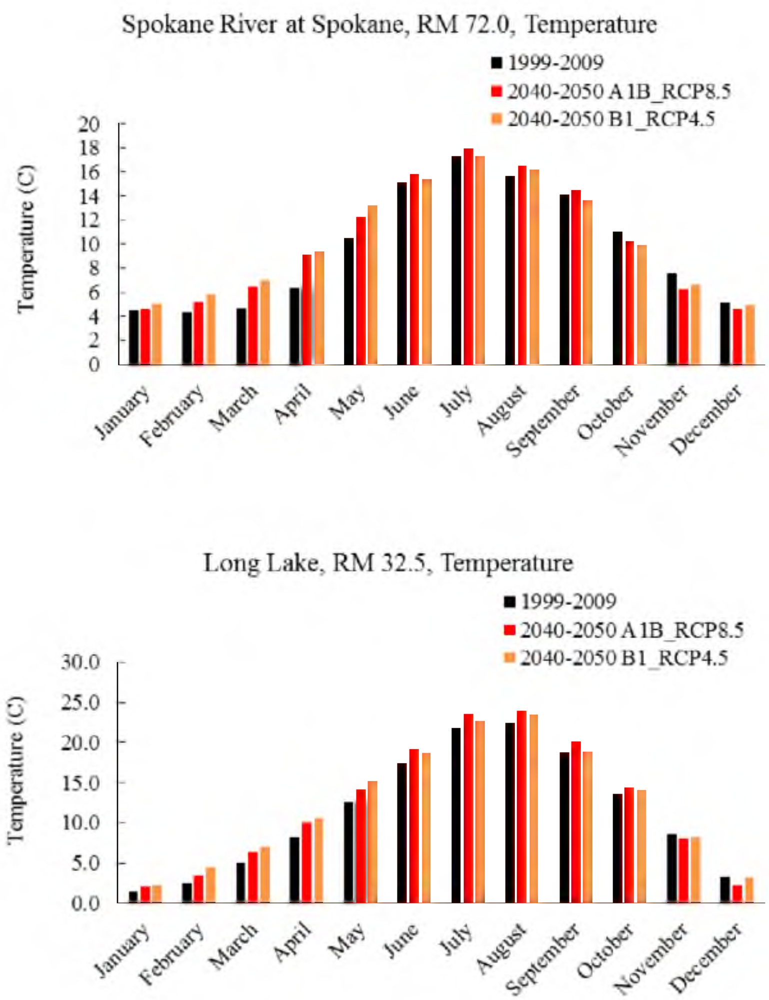


Figure 4.18 Comparison of Climate Scenario Temperature Results at Monthly Scale

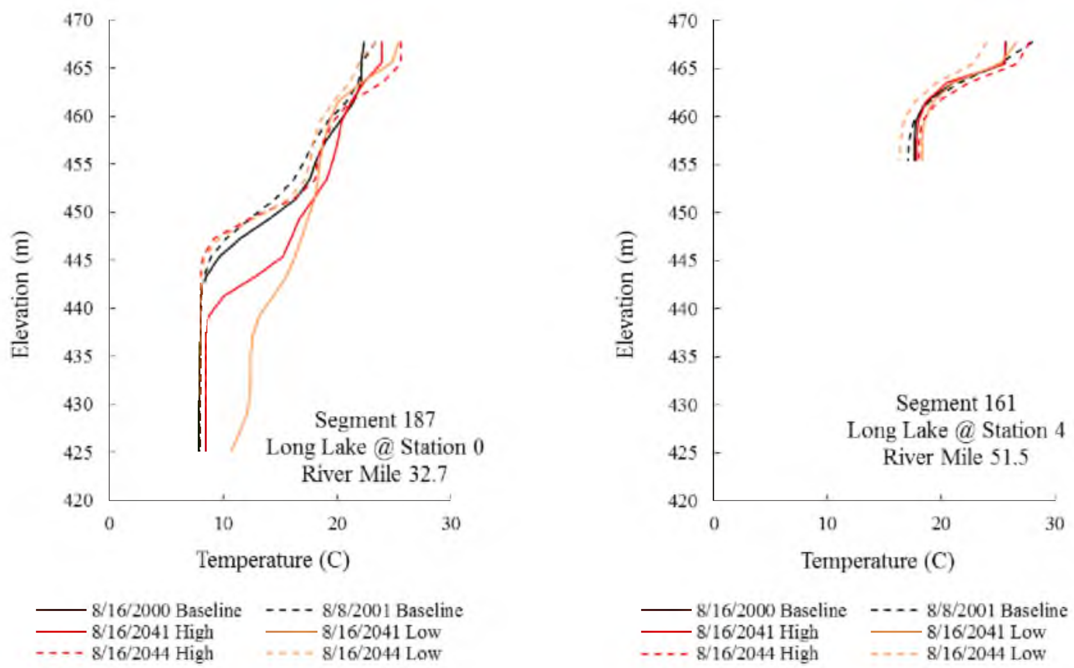


Figure 4.19 Temperature Vertical Profiles Comparing Baseline and Climate Scenario

4.5.1.3 Dissolved Oxygen

Projected dissolved oxygen concentrations (at surface) at the Spokane River at Spokane and Long Lake sites during 2040-2050 are shown in Figure 4.20. The basic dissolved oxygen seasonal pattern seems to persist in the projected climates. Although DO at Spokane River at Spokane seemed to agree well with the minimum criteria (8 mg/L), Long Lake DO concentrations showed regular violation of the criterion during summer months. This is possibly due to the higher water temperatures lowering the capacity of water to hold oxygen (Solheim et al., 2010; Chang et al., 2015). The decreased dissolved oxygen might also be the result of higher algal growth (Solheim et al., 2010; Paerl and Paul, 2012; Lee et al., 2012). The swing between high and low DO concentrations at Long Lake changed from 6.3–14.0 mg/L to 5.5–15.4 mg/L in the modified climates compared to

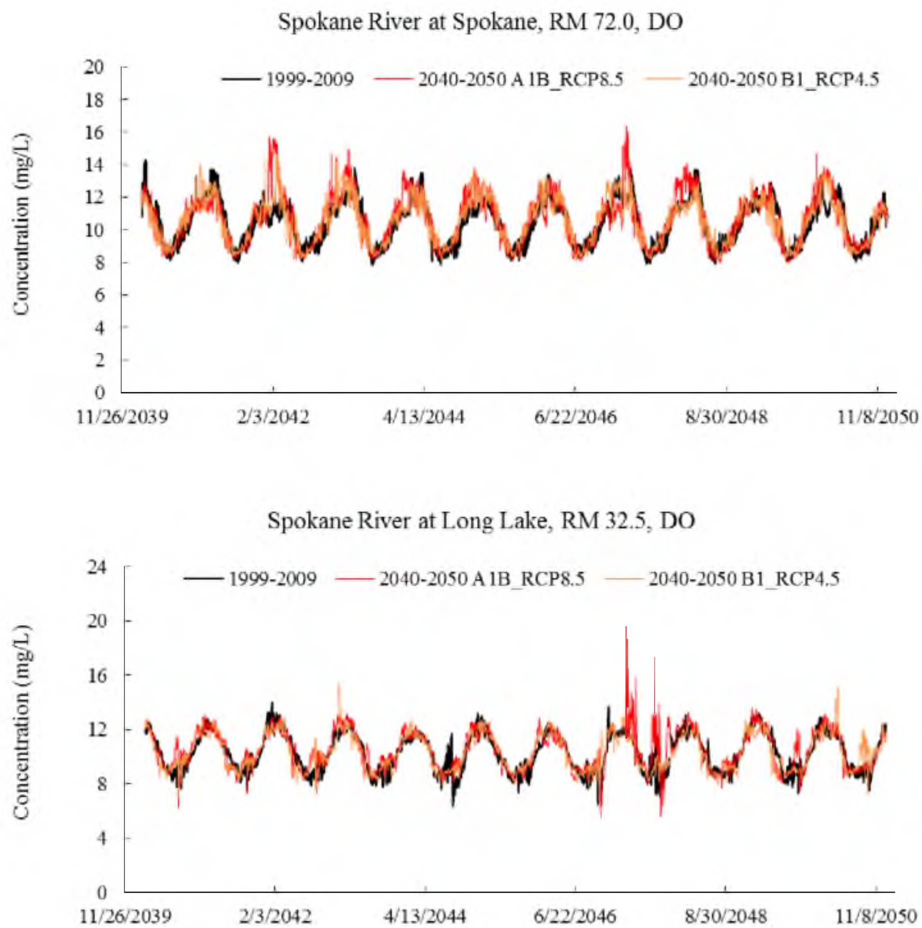


Figure 4.20 Climate Change Model Simulation Results for Dissolved Oxygen

baseline scenario. This is possibly due to the effect of higher algal presence. Analysis of DO for other sites revealed violation of criteria for river segments 67 to 86 (RM 79.7 – 74.8), and reservoir segments 174 to 188 (RM 42.1 – 32.7) during summer to late summer.

Monthly analysis of dissolved oxygen for the Spokane River at Spokane and Long Lake sites, shown in Figure 4.21, revealed that the 2040-2050 average concentrations during late summer were very close to the DO standard. Statistical analysis showed that the dissolved oxygen concentration differences between baseline and climate scenarios at

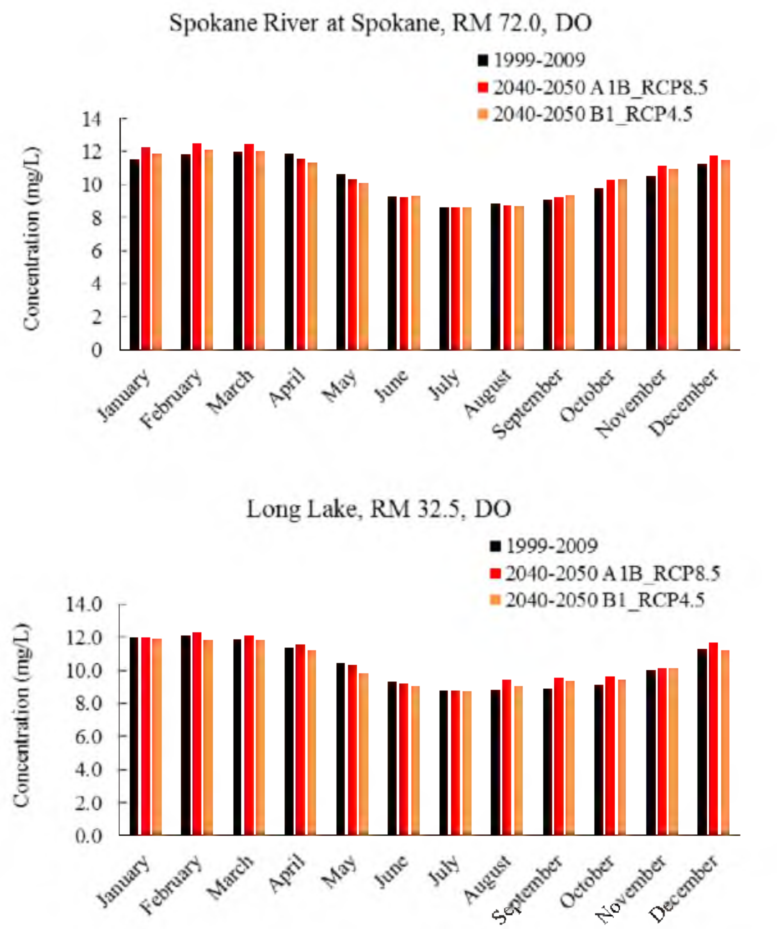


Figure 4.21 Comparison of Climate Scenario DO Results at Monthly Scale

different river locations were statistically significant (0.05 significance level, $p < 0.005$), but not for concentrations at Long Lake ($p > 0.1$). This gives an impression that the issue with low DO at Long Lake during summer might still persist in future times. Oxygen at depth seemed to suffer as well (Figure 4.22), presumably due to increased temperature reducing oxygen flux from the atmosphere (Golosov et al., 2012; Chang et al., 2015) and higher algal activity (Cox and Whitehead, 2009; Paerl and Paul, 2012). Vertical profiles at Segment 187 showed severe violation of the water quality criteria set for DO.

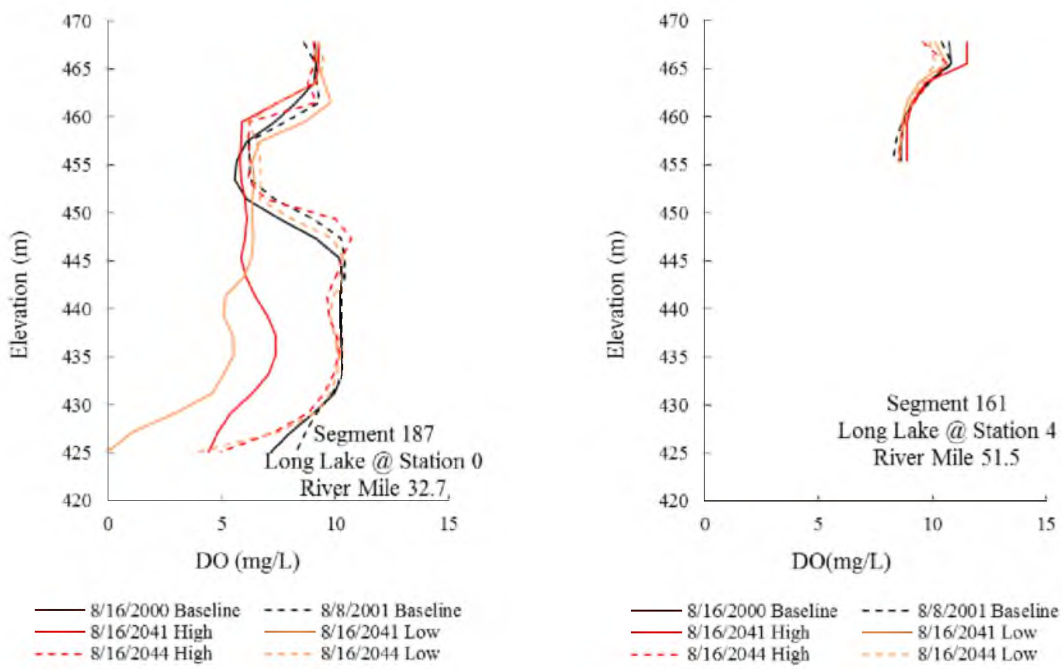


Figure 4.22 DO Vertical Profiles Comparing Baseline and Climate Scenario

4.5.1.4 Phosphate-Phosphorous

Projected phosphate concentrations (at surface) during 2040-2050 at Spokane River at Spokane and Long Lake are shown in Figure 4.23. Phosphate concentrations for both low and high emission scenarios were lower than the baseline scenario at the river sites. This most likely occurred due to the dilution effect from increased streamflow (Anthony et al., 2004; Ludwig et al., 2009). The lower phosphate concentrations might also be attributed to higher algal uptake. However, at Long Lake, concentrations exceeded the water quality criterion for phosphate (0.25 mg/L) in summer during the 2040-2050 period.

On an average, the projected phosphate concentrations for high and low emission scenarios were 0.001 mg/L lower than the baseline scenario at river sites, while Long Lake concentrations were 0.004 mg/L lower than the baseline scenario. Overall, concentrations

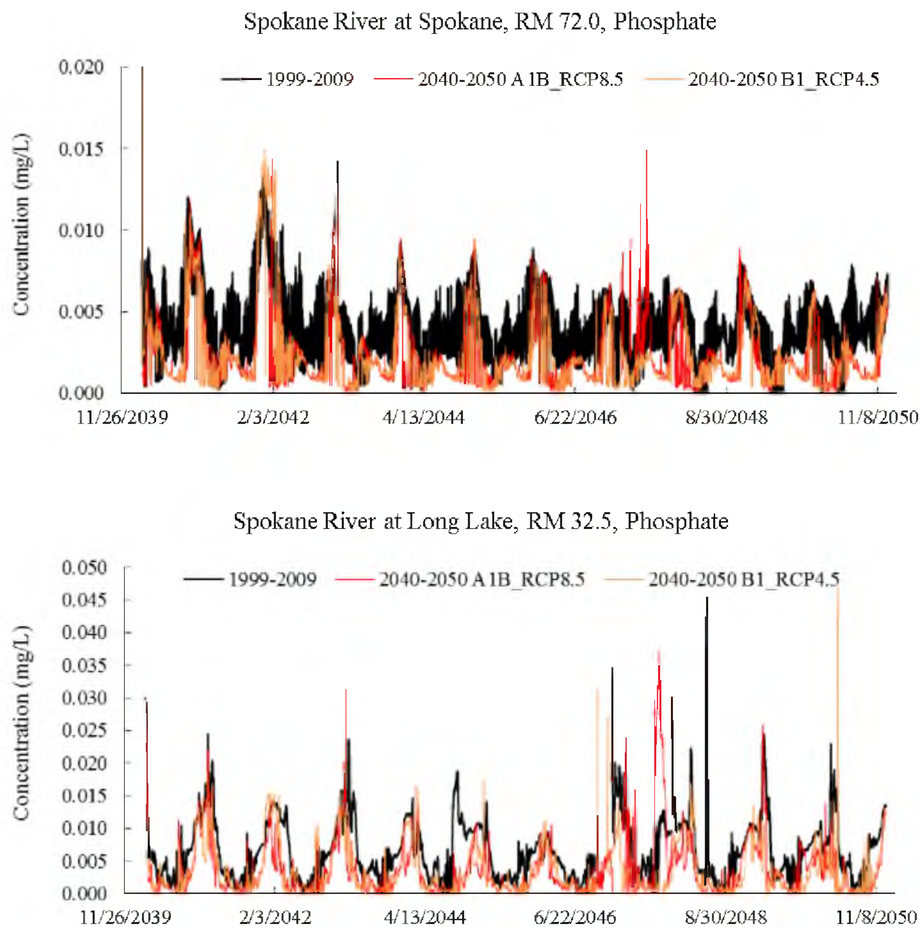


Figure 4.23 Climate Change Model Simulation Results for Phosphate

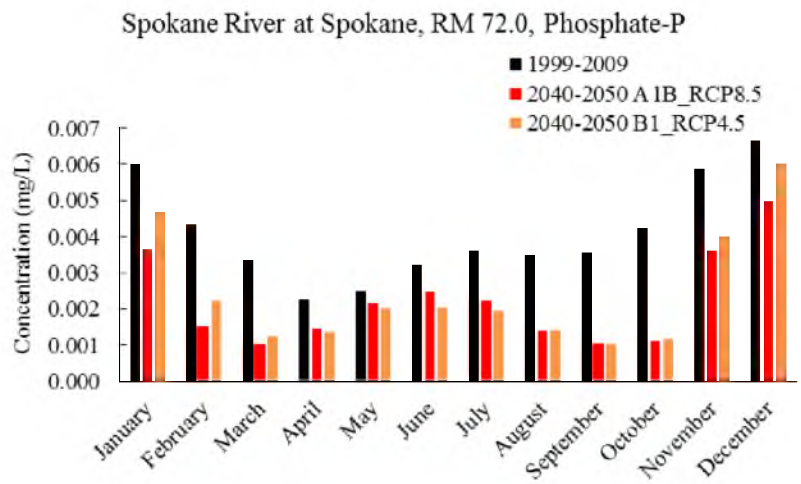
decreased by 25-30% under the climate change impacts. However, an increasing trend in concentrations was seen from Stateline towards Long Lake (analysis not shown), indicating degraded conditions at downstream locations.

Two-tailed paired t-test showed that the surface phosphate concentration differences between the baseline and climate scenarios at river locations were statistically significant (0.05 significance level, $p < 0.001$), which can be attributed to the dilution effect.

Figure 4.24 shows the projected monthly phosphate concentrations at Spokane River at Spokane and Long Lake sites. Concentrations at the Long Lake site were lowest during summer (April to August), while highest during winter (December to March). This occurred from the reduction of streamflows in summer and subsequent decrease in sediment loadings (Chang et al., 2001). For Spokane River at Spokane site, phosphate concentrations were highest during winter (November to January) but decreased during spring and then moderately increased in summer (May to July). Higher streamflows in winter were responsible for the concentration increase in winter (Chang et al., 2001). Overall, the projected concentrations were lower than the baseline scenario for all months. The effect of dilution was also apparent from the vertical profiles, shown in Figure 4.25. However, there was evidence of phosphate concentration criteria violation at deeper layers of Long Lake (Segment 187) for a relatively low flow year (2041). Release of phosphorus from sediment due to enhanced anoxia in deep layers (Chang et al., 2015) can worsen this state.

4.5.1.5 Ammonia-Nitrogen

Projected ammonia-nitrogen concentrations (at surface) at Spokane River at Spokane and Long Lake sites during 2040-2050 are shown in Figure 4.26. Ammonia-nitrogen surface concentrations for both low and high emission scenarios were lower than the baseline scenario for the Spokane River site. This can be attributed to the dilution effect from increased streamflow (Anthony et al., 2004; Ludwig et al., 2009). Concentrations, in general, decreased from upstream to downstream locations. For Long Lake site, the concentrations looked more like the baseline values with occasional peaks in summer.



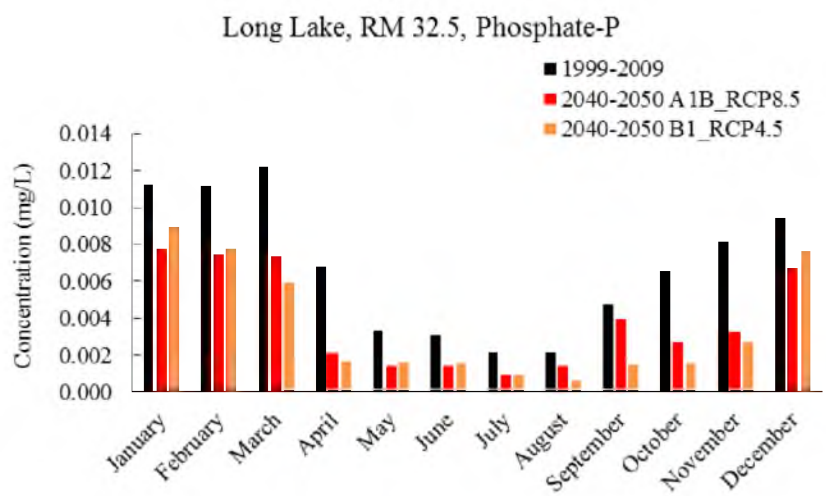


Figure 4.24 Comparison of Climate Scenario Phosphate Results at Monthly Scale

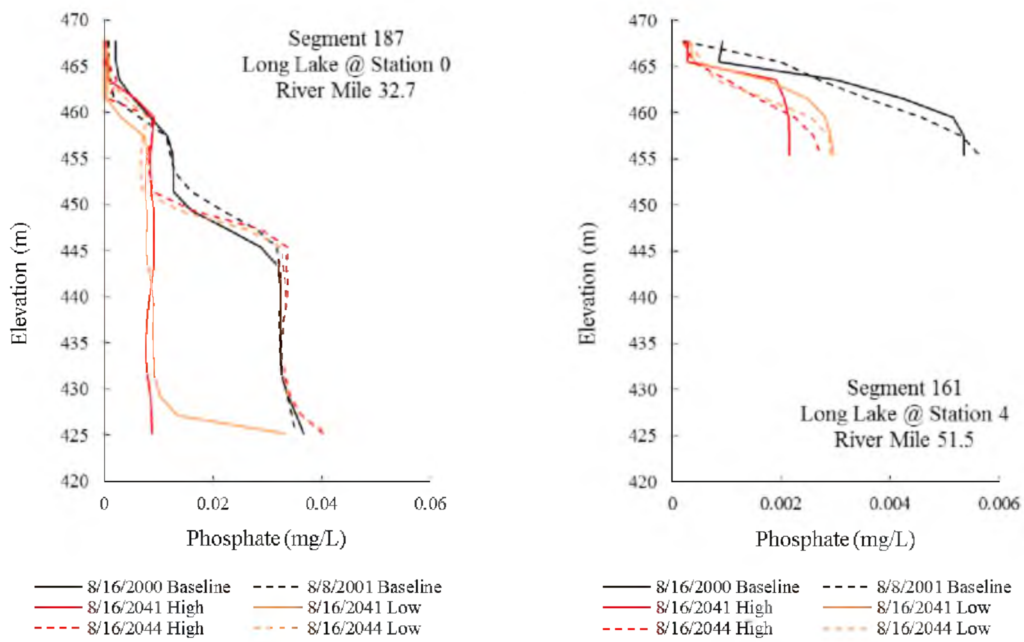


Figure 4.25 Phosphate Vertical Profiles Comparing Baseline and Climate Scenario

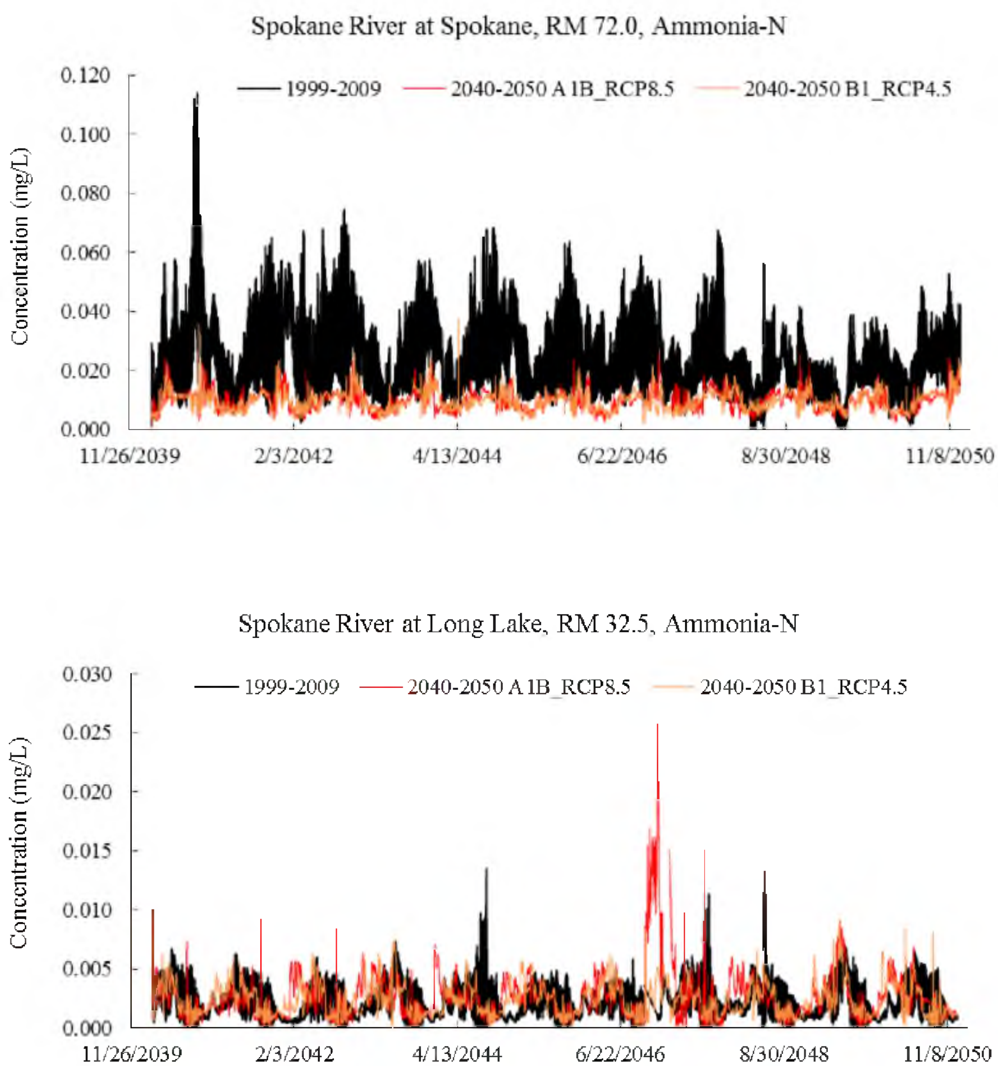


Figure 4.26 Climate Change Model Simulation Results for Ammonia-Nitrogen

On an average, the ammonia-nitrogen concentrations were 0.013 mg/L lower than the baseline scenario for river sites. The difference in projected and baseline concentrations was greater for sites from Sandifer Bridge to Nine Mile (from analysis). A two-tailed paired t-test showed that the ammonia-nitrogen concentrations differences between the baseline and climate scenarios at river locations were statistically significant (0.05 significance level, $p < 0.001$). Figure 4.27 shows the monthly analysis of ammonia-nitrogen concentrations at Spokane River at Spokane and Long Lake.

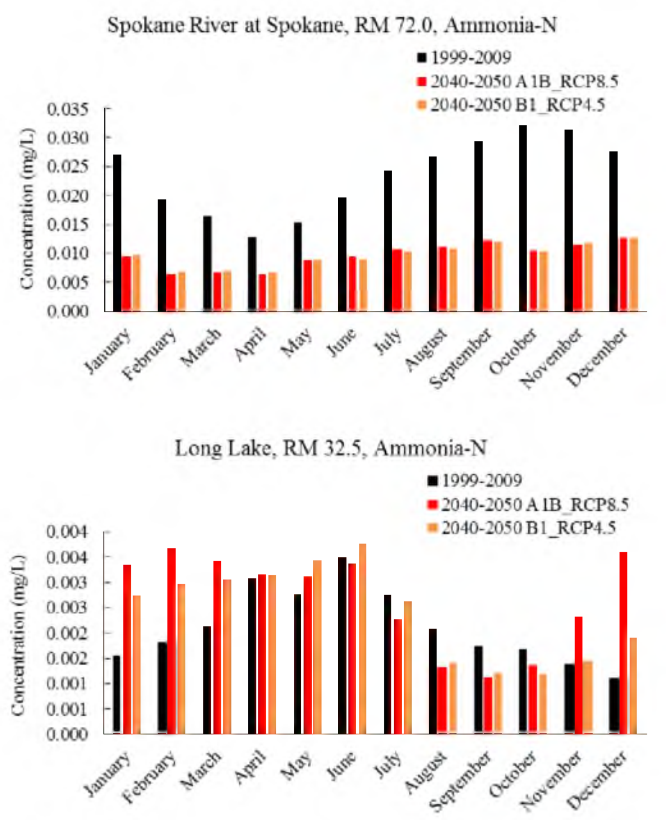


Figure 4.27 Comparison of Climate Scenario NH_3-N Results at Monthly Scale

For the Spokane River site, projected concentrations were lower than the baseline scenario for all months. But at Long Lake, projected concentrations were higher than the baseline from November-May. This can be attributed to the increased loading carried by higher streamflows in winter (Chang et al., 2001). July-October showed a decrease in projected concentrations. Figure 4.28 shows the vertical profiles of ammonia-nitrogen concentrations for climate scenarios, where somewhat mixed impacts are visible. Higher concentration near the bottom may have been the result of ammonification due to oxygen depleted conditions (Lillebø et al., 2007).

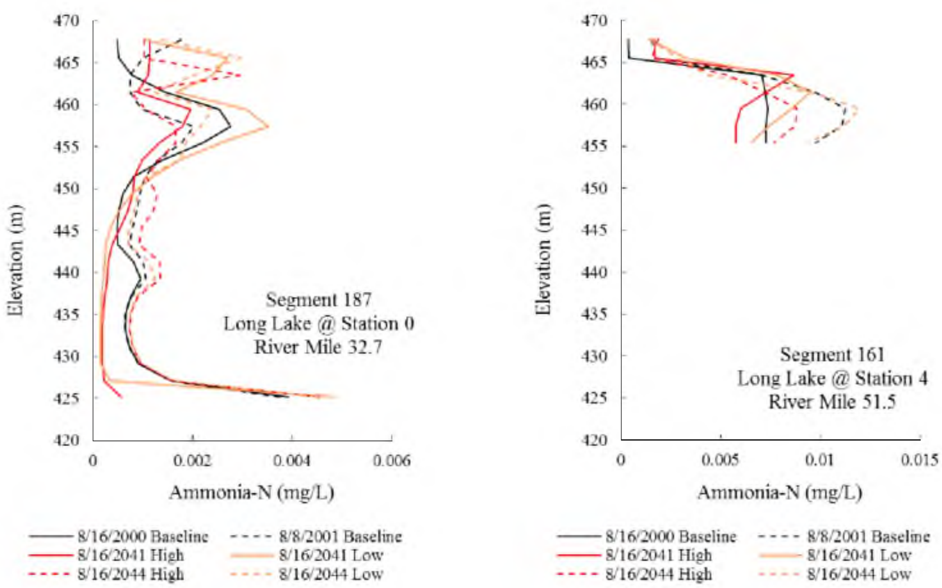


Figure 4.28 Ammonia-N Vertical Profiles Comparing Baseline and Climate Scenario

4.5.1.6 Nitrate-Nitrogen

Projected nitrate-nitrogen surface concentrations at Spokane River at Spokane and Long Lake sites are shown in Figure 4.29. Projected nitrate-nitrogen concentrations for both high and low emission scenarios during 2040-2050 were similar to the baseline scenario, with slight decrease in average values. This was perhaps due to the dilution occurring from increased streamflow (Anthony et al., 2004; Ludwig et al., 2009). The lower nitrate concentrations might also be attributed to the increased algal uptake (Lee et al., 2012) and higher denitrification rate (Chang et al., 2015) due to increased water temperatures.

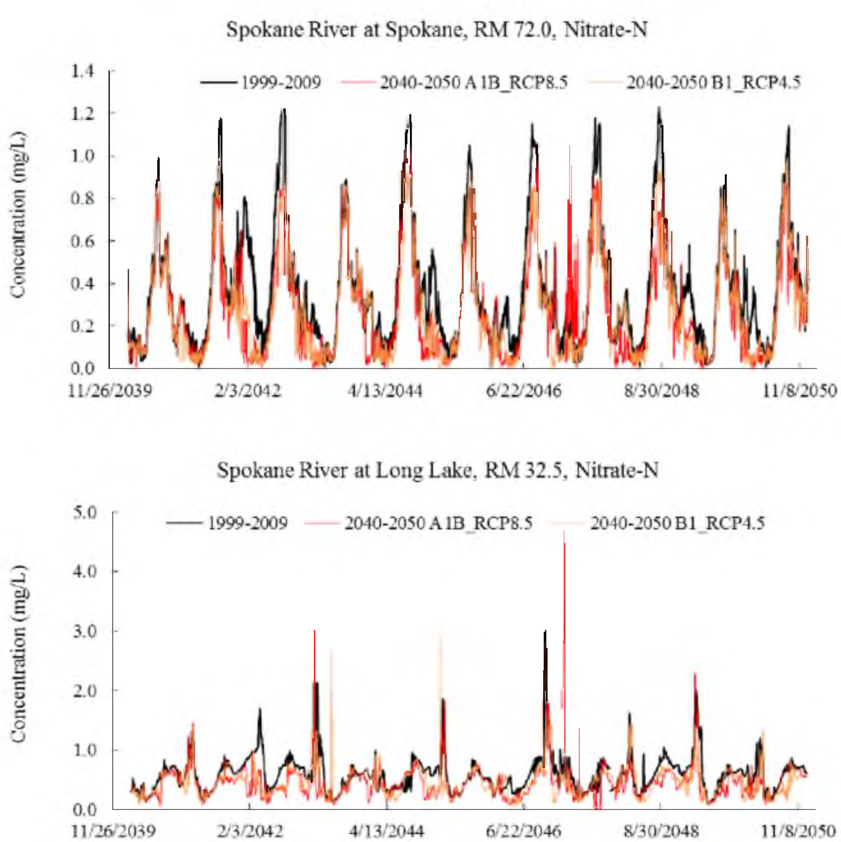


Figure 4.29 Climate Change Model Simulation Results for Nitrate-Nitrogen

For the river sites, the projected nitrate-nitrogen concentrations showed a decrease of approximately 0.06 mg/L, while concentrations at Long Lake site showed a decrease of about 0.16 mg/L on average. Overall, projected concentrations were 20% lower than baseline values. However, an increasing trend in concentrations was seen from upstream to downstream locations (analysis not shown), indicating degraded conditions with proximity to Long Lake. Two-tailed paired t-test showed that the nitrate-nitrogen concentration difference between the baseline and climate scenarios (high and low emissions) at the river sites and Long Lake were statistically significant (0.05 significance level, $p < 0.001$). Figure 4.30 shows the comparison of monthly nitrate-nitrogen concentration at Spokane River at Spokane and Long Lake.

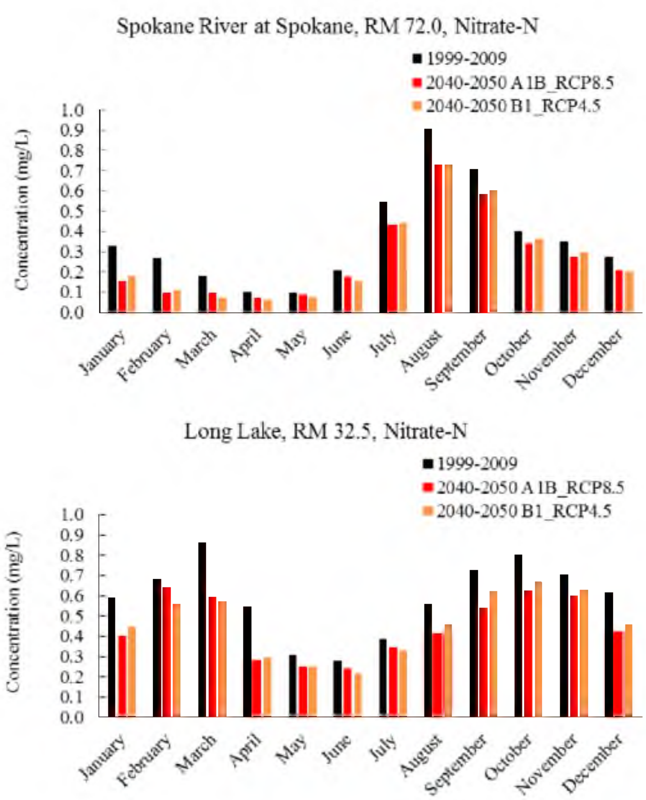


Figure 4.30 Comparison of Climate Scenario Nitrate-Nitrogen Results at Monthly Scale

Lowest nitrate concentrations were seen during winter months at the Spokane River site, while summer months had the highest concentrations. This was most likely due to the lower nitrification rates at the river sites during winter (Verweij et al., 2010). On the contrary, at Long Lake, summer months had the lowest nitrate-nitrogen concentrations, perhaps due to higher algal uptake and microbial denitrification rates (Bark, 2010; Chang et al., 2015). Nonetheless, nitrate-nitrogen concentrations for both high and low emission climate scenarios were lower than the baseline for all months. Figure 4.31 shows the vertical profiles of nitrate-nitrogen concentrations for climate scenarios, where the effect of dilution from increased streamflows was noticeable.

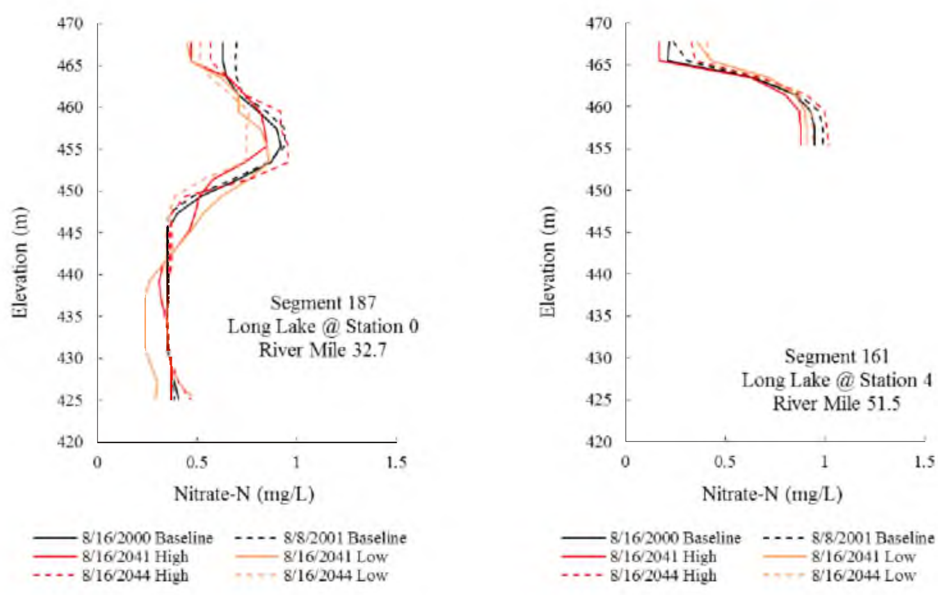


Figure 4.31 Nitrate-Nitrogen Vertical Profiles Comparing Baseline and Climate Scenario

4.5.1.7 Total Algae

Figure 4.32 shows the projected total algae concentrations at surface for 2040-2050 at Spokane River at Spokane and Long Lake sites. For both sites, total algae concentration showed significant increase. This was mostly due to the increase in water temperature, which favored algae growth (Arvola et al., 2010; Moore and Ross, 2010). Algal concentrations of different groups were analyzed, and cyanobacteria was found to dominate the algae community due to its affinity for higher temperatures (Solheim et al., 2010; Nõges et al., 2010; Paerl and Paul, 2012).

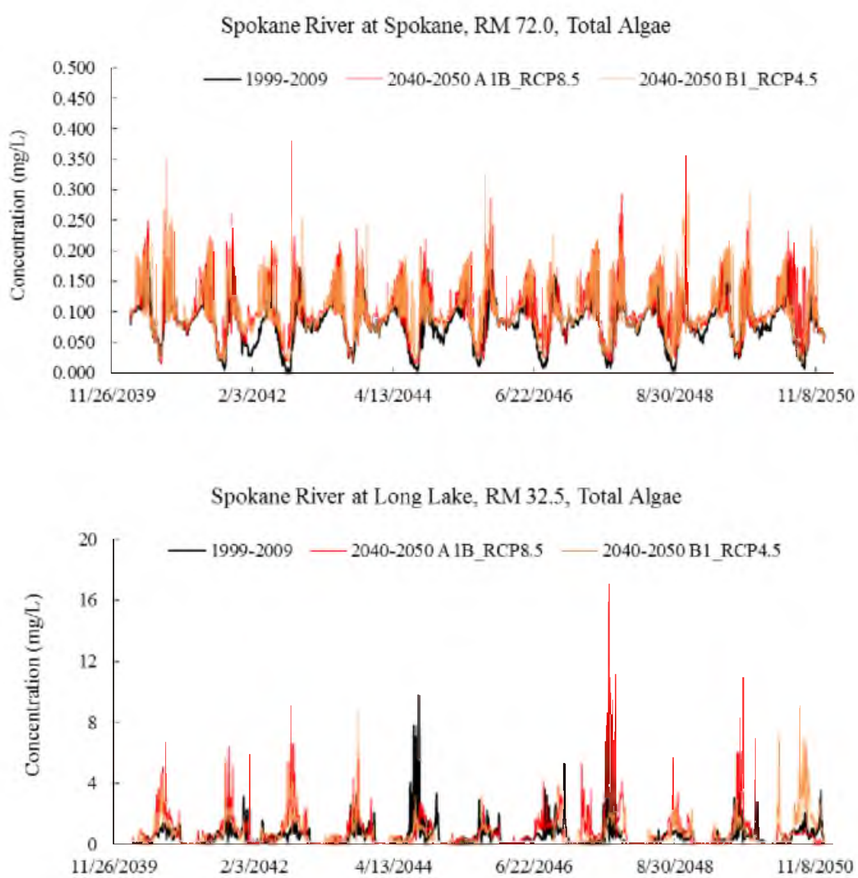


Figure 4.32 Climate Change Model Simulation Results for Total Algae

Total nitrogen (TN) to total phosphorous (TP) ratio (TN/TP) in the climate change scenarios decreased by approximately 20% compared to the baseline scenario. This decrease can mean either a nitrogen deficiency or an overabundance of phosphorus (Tong et al., 2007).

Comparison of total nitrogen and total phosphorous concentrations from climate scenario simulations with baseline scenario indicated that the system suffered from an overabundance of phosphorus. Relative increase of TP in climate scenarios might also have supported algae growth. Higher algae growth was also responsible for wider dissolved oxygen swings (low to high).

Figure 4.33 shows the comparison of monthly total algae concentrations at Spokane River at Spokane and Long Lake sites, which indicated that the monthly pattern is persistent in the climate scenarios. For the Spokane River at Spokane site, algae concentrations seemed to be lowest is August. For Long Lake, concentrations were highest during late summer, when their activity peaks (Moore and Ross, 2010).

Overall, the projected algae concentrations were higher than baseline values for all months at river sites and Long Lake. Spatially, the algae concentrations were found to decrease from Stateline (RM 96.0) to Nine Mile (RM 58.0) location (analysis not shown), but increased drastically at Long Lake sites (RM 33.0).

Figure 4.34 compares the vertical profiles of total algae concentrations for high and low emission climate scenarios with the baseline scenario, where the impact of increased temperature affecting higher algal growth (Moore and Ross, 2010; Arvola et al., 2010) was noticeable.

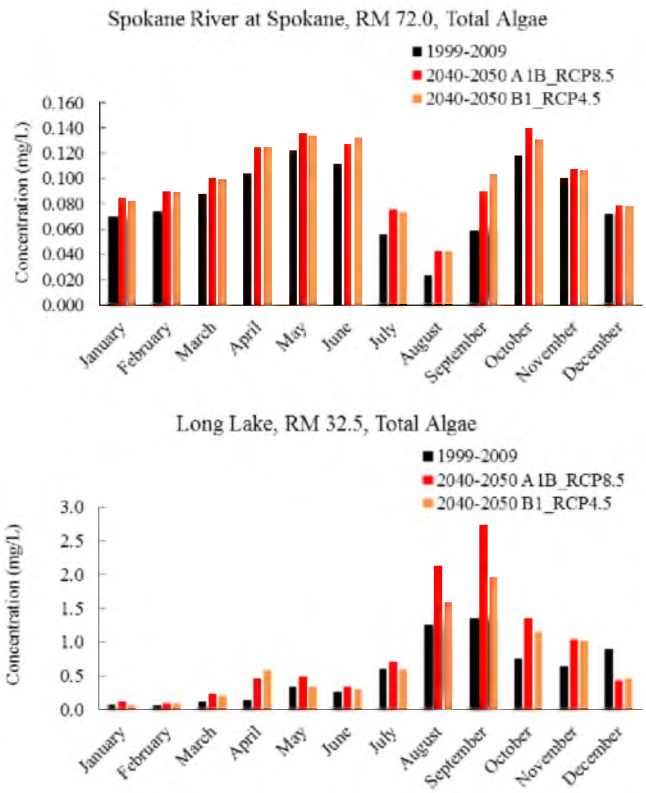


Figure 4.33 Comparison of Climate Scenario Total Algae Results at Monthly Scale

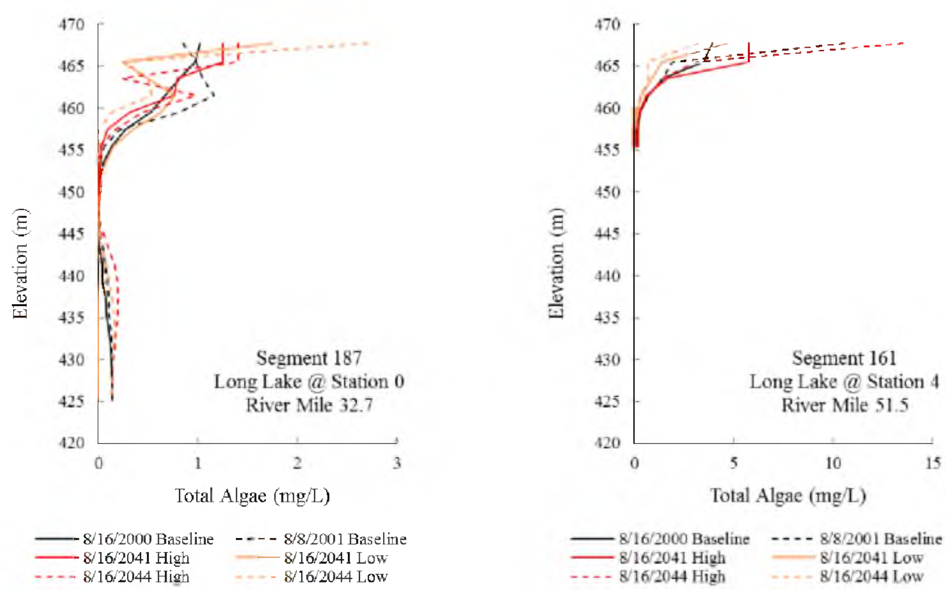


Figure 4.34 Total Algae Vertical Profiles Comparing Baseline and Climate Scenario

4.5.2 Alternative Scenario Evaluation

An alternative scenario to assess the effect of point and nonpoint source loading reduction under the climate change and population growth impacts was simulated. The aim was to observe how effective a 90% point load and 50% nonpoint load reductions was in terms of water quality improvement in the Spokane River-Long Lake system. From the plots in Figure 4.35, showing the effect of point and nonpoint source load reduction on Long Lake water quality under the climate change and population growth scenarios, it was seen that the source reduction can be effective in reducing the surface nutrient concentrations to some extent, but the problem with low dissolved oxygen still persists. Dissolved oxygen continued to go well below the 8 mg/L standard, especially during summer, due to the respiration of the organic matter produced from primary production. Additionally, contribution ratios of nutrient loads from nonpoint loading increased as streamflow increased (Du et al., 2014). Model results indicated the need for higher load removal/control from nonpoint sources, particularly during winter high flows, to prevent low dissolved oxygen in the Long Lake-Spokane River system.

From a seasonal point of view, the consequence of point and nonpoint source load reduction seemed to have some positive effect (in terms of nutrient concentrations at Long Lake) under climate change impacts, as seen from Figure 4.36. However, relatively lower phosphate and nitrate-nitrogen concentrations during summer indicated that most of the nutrient loading occurred during the winter and spring period when flows were higher. Thus summer, which is typically considered the critical time of the year in terms of water quality, does not pose the only cause of concern.

Although reduction of point and nonpoint source loading under climate change

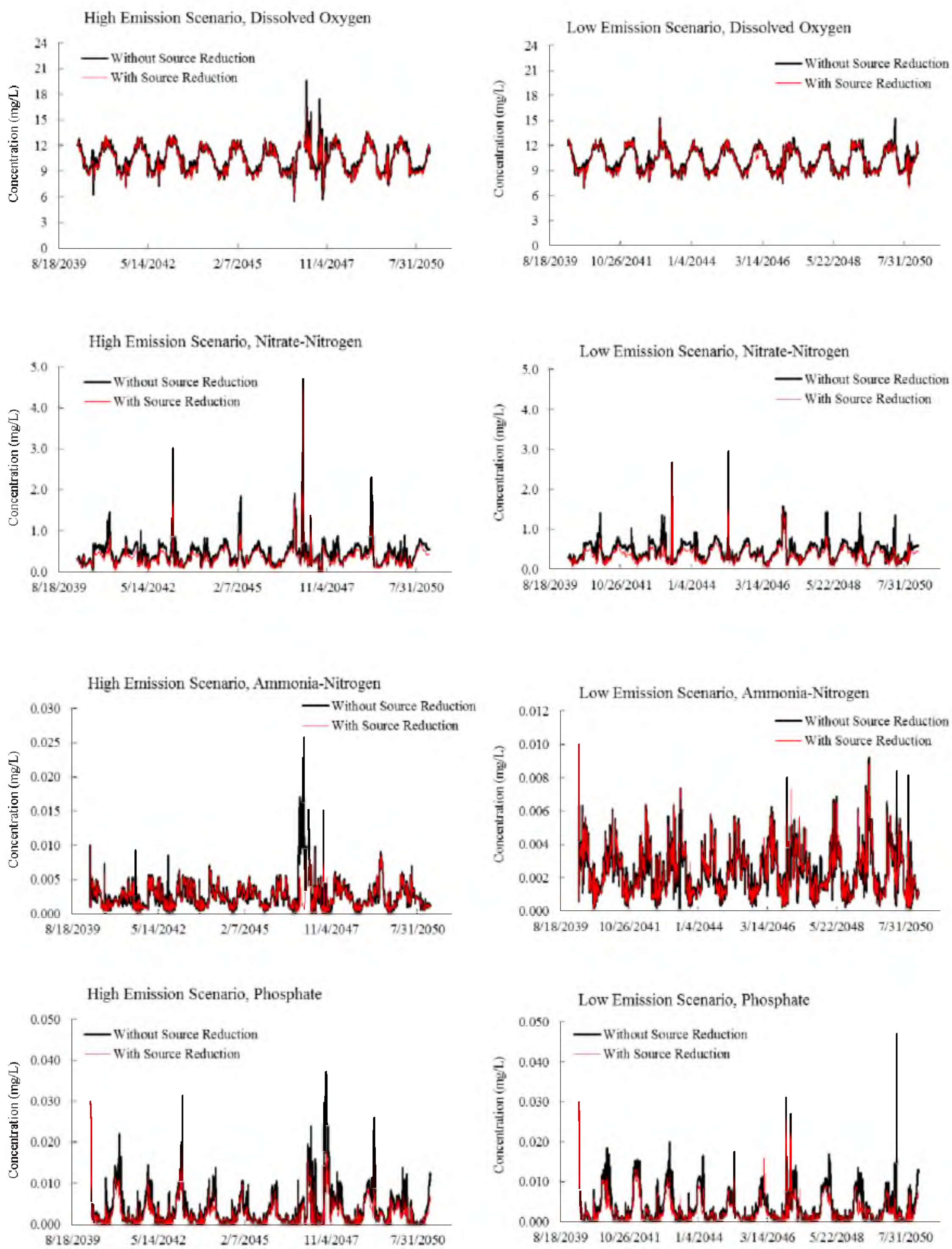


Figure 4.35 Effect of Source Loading Reduction in Climate Change Scenarios, Long Lake

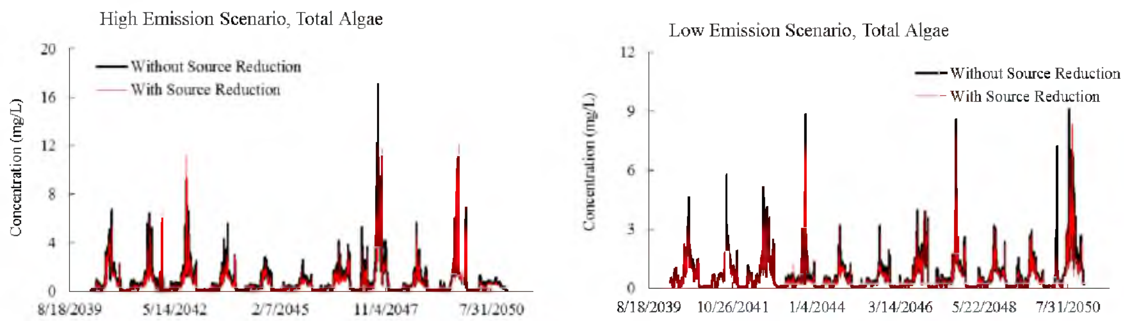


Figure 4.35 Continued

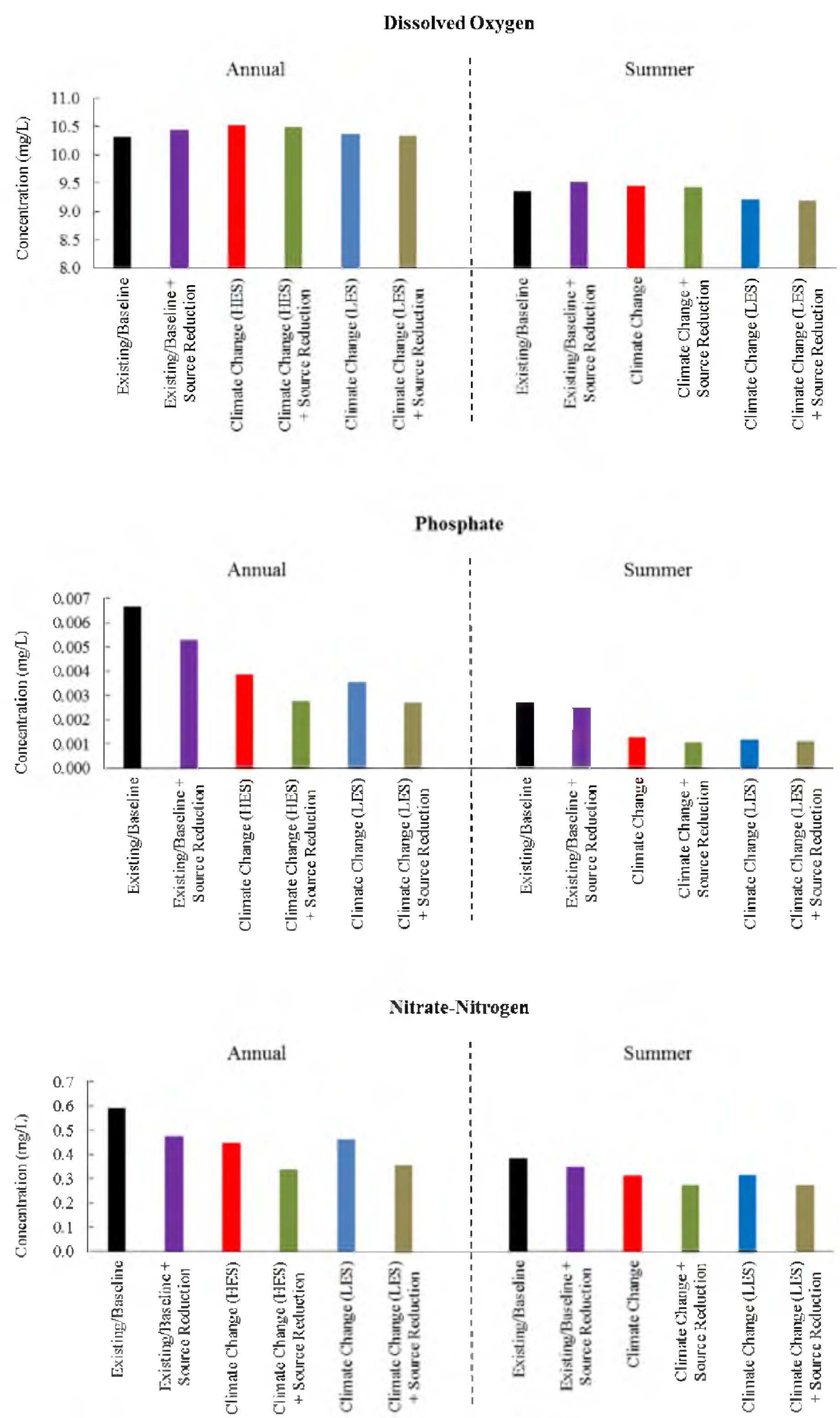


Figure 4.36 Impact of Point and Nonpoint Loading Reduction on Water Quality at Long Lake (HES: High Emission Scenario, LES: Low Emission Scenario)

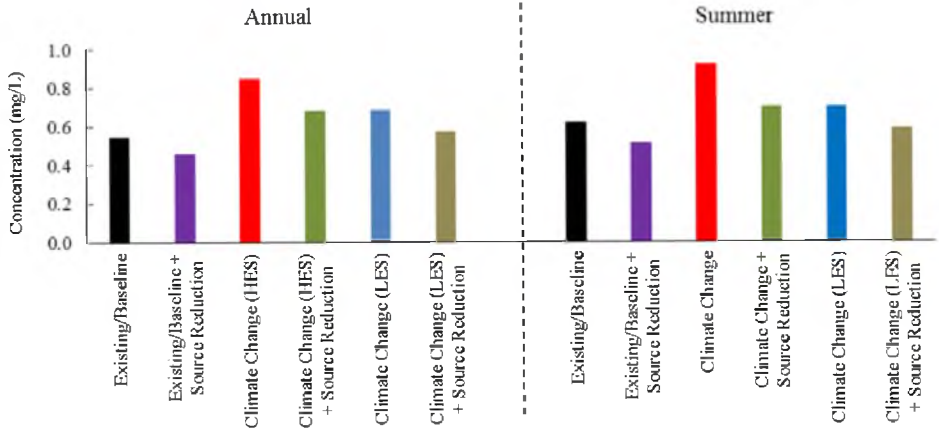
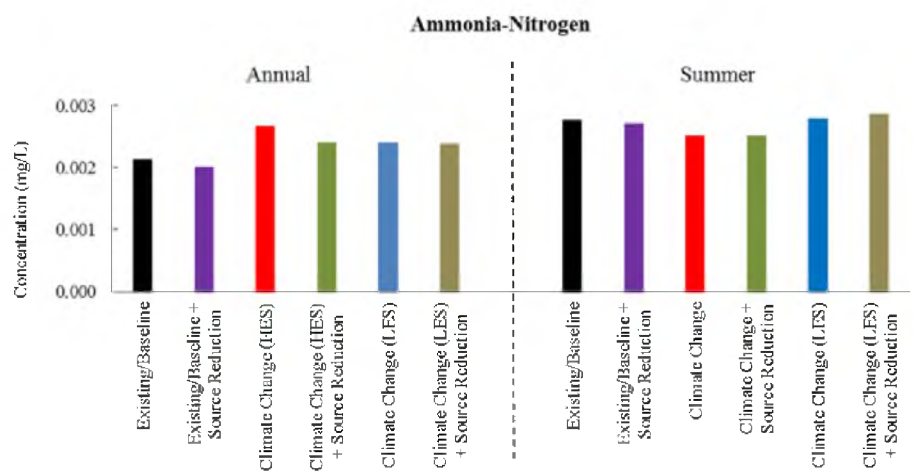


Figure 4.36 Continued


Ammonia-Nitrogen
Annual
Summer
Concentration (mg/L)
0.003
0.002
0.001
0.000
Existing/Baseline
Existing/Baseline + Source Reduction
Climate Change (HES)
Climate Change (HES) + Source Reduction
Climate Change (LES)
Climate Change (LES) + Source Reduction
Existing/Baseline
Existing/Baseline + Source Reduction
Climate Change
Climate Change + Source Reduction
Climate Change (LES)
Climate Change (LES) + Source Reduction

Total Algae

impacts appeared to have improved the water quality of surface layers, nutrient and dissolved oxygen concentrations in the deeper layers of the Long Lake did not improve much. As seen from Figure 4.37, concentrations at deeper layers of the Long Lake reservoir hardly saw any change during the critical late summer period upon load reductions. Dissolved oxygen and phosphate-phosphorous concentrations in the deeper layers still violated their respective criteria.

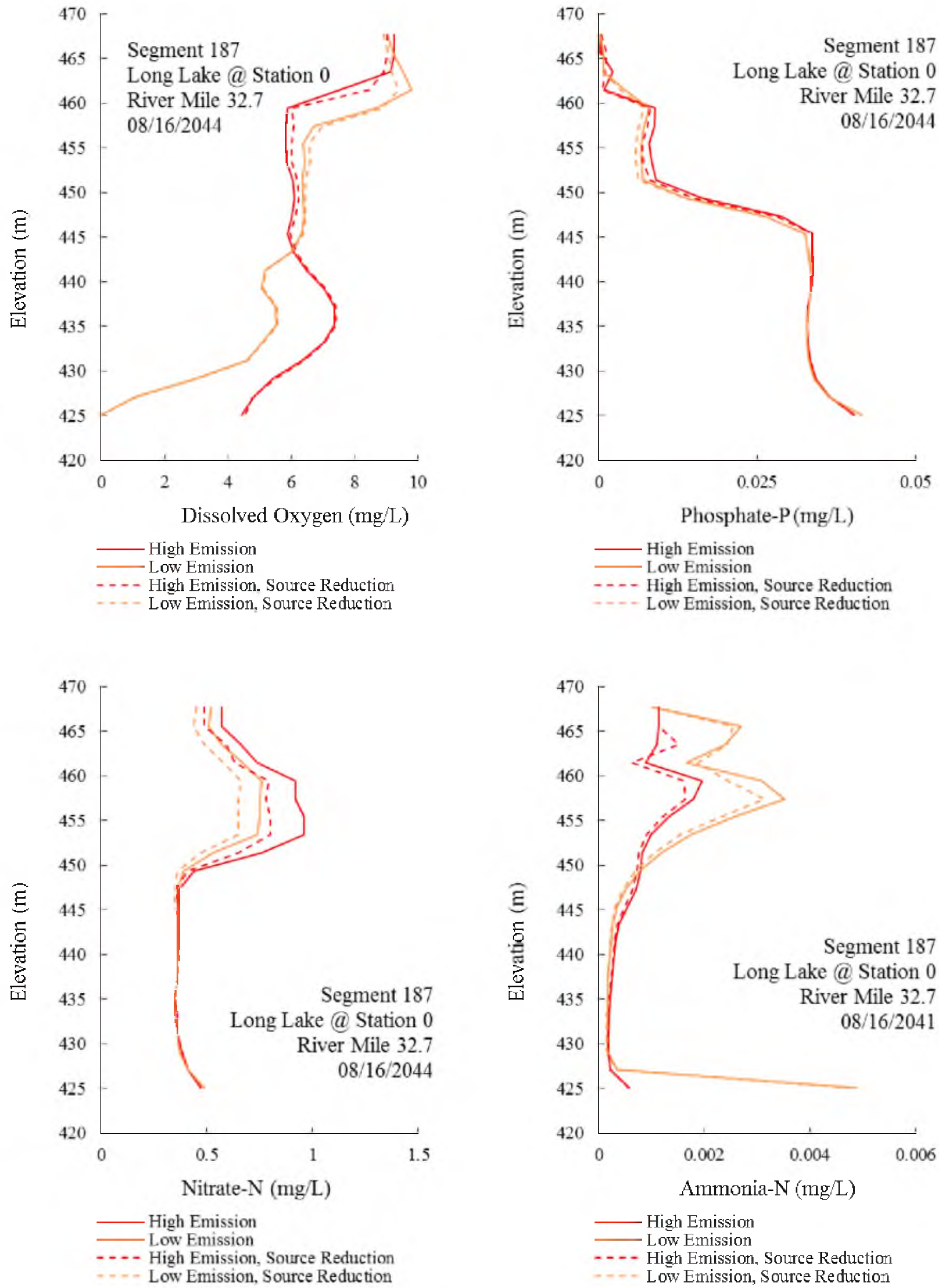


Figure 4.37 Impact of Source Loading Reduction on Concentration Profiles, Long Lake (Segment 187)

CHAPTER 5

DISCUSSION OF RESULTS

5.1 Realization of Objectives

The primary goal of this project was to improve the understanding of how changes in watershed conditions due to population growth and climate change may alter management decisions with respect to wasteload and load allocation in the TMDL process. The objectives included expanding and calibrating an existing low flow TMDL model to encompass varying hydrologic condition that includes the impacts of stormwater runoff; evaluate how long-term simulations compared to low flow analysis in terms of nutrient reduction requirements; examine the impacts of population projections on point and nonpoint loads under climate induced flow conditions; and investigate the impacts of climate change on decisions related to point and nonpoint source nutrient loading reduction.

The first objective of the study was achieved by extending the 2001 CE-QUAL-W2 (Version 3.1) Spokane River-Long Lake model to a period 1999-2009 and calibrating it to field data. In general, the model satisfactorily reproduced the river and reservoir responses to known boundary conditions. Calibration errors within acceptable limit justified the suitability of the model for evaluating the impacts of climate change and population growth on Spokane River - Long Lake water quality and management strategies in sustaining the water quality.

Comparison of long-term model simulations to the low flow analysis, the second objective of the study, revealed that hydrologic conditions outside the low flow period may be cause for concern, and perhaps the impact of nonpoint source loading on nutrient cycling still needs better understanding. The contribution ratios of nutrient loads from nonpoint loading increased as streamflow increased, a fact that did not become apparent from the low flow analysis. The load reduction requirements from point and nonpoint sources under the existing TMDL was found to be inadequate in supporting water quality standards over long term. Several instances of violation of dissolved oxygen criteria were observed during the historic time period (1999-2009), both with and without source loading control; which does not fulfil the Spokane River–Long Lake TMDL project's primary goal of meeting Washington State's water quality standard for dissolved oxygen. Model results suggest the need for significant reduction of nonpoint source loading in order to prevent low dissolved oxygen in the river-lake system.

Simulation of the impact of population projections on point loads under climate induced flow conditions, the third objective of the study, helped understand the extreme nature of climate change impacts. Disproportionate increase of streamflows during winter, as found from this study, can overwhelm the existing stormwater flow and pollution control infrastructure through exceedingly high nutrient loadings. This simply adds to the existing problem with low summer flows in the Spokane River, as excess nutrient loads from high flow winter season may have consequences on water quality during summer. Although the increase in streamflow due to climate change impacts seemed to have some positive effect on water quality at surface layers because of dilution, nutrient and dissolved oxygen concentrations in the deeper layers did not experience much improvement. Water quality in the river, in terms of dissolved oxygen and nutrient concentrations, degraded from upstream to downstream locations. Model results with modified climate showed violation of dissolved oxygen criteria at river locations – Upriver Dam, Green Street Bridge, and Walkbridge behind Spokane Center (RM 79.7 to RM 74.8); and reservoir locations – Long Lake Station 0, 1 and 2 (RM 42.1 to RM 32.7), particularly during summer to late summer (July to October). Controlling nonpoint source pollution in the Spokane River and Long Lake, particularly during high flows, appeared to be a necessity in future time periods to maintain water quality. In addition, improved nutrient removal from wastewater treatment plants will contribute to sustaining river-lake water quality.

While investigating the impacts of climate change on management decisions related to point and nonpoint source nutrient loading reduction, the fourth objective of the study, it was seen that the loading reduction requirements under the existing dissolved oxygen TMDL, although it looked satisfactory in maintaining water quality standards at surface layers, was not adequate in sustaining the oxygen and nutrient concentrations at the bottom. Such multifaceted nature of climate change effects on water quality makes management decisions more complex for water managers. This even creates doubt on the point and nonpoint source loading reduction citations in the existing TMDL (based on low flow analysis) targeted towards meeting dissolved oxygen standards in the Spokane River-Long Lake system. Results implied that the target set for wasteload and load allocations and the associated loading reduction may need to be revised in order to meet water quality standards under the climate change and population growth impacts.

5.2 Recommendations

Changes in the timing and the magnitude of streamflow and associated dissolved oxygen and nutrient concentrations have implications for water quality management as well as aquatic ecosystems in the Spokane River-Long Lake system. The likely effects of the peak nutrient concentrations are especially of interest because of their effects on algal blooms in the Spokane River-Long Lake system and the requirement for dischargers to meet water quality standards.

Based on this study, there are several ways in which the modeling approach may be improved for less problematic assessment of potential climate change effects in the Spokane River and Long Lake. For example, in this study, some of the water quality data required were not available, and had to be estimated. Groundwater data were scarce as well. Zooplankton was not simulated in the model under the assumption that it does not have a significant impact on the algal dynamics and nutrient recycling. Also, the model currently assumes one phytoplankton species to be sufficient for representing the overall primary production and nutrient interactions in the system. Hence, more intensive historical water quality data compiled over longer time periods, particularly the boundary conditions and groundwater flows, would possibly help in minimizing the discrepancies between observed and predicted dissolved oxygen and nutrient concentrations. Additional vertical profile data would also result in a better understanding of the model system. It may also be useful to change some of the parameters that were held constant during the future climate scenario simulations in this study. While projecting the future changes in water quality, only the climate inputs (temperature, dew point, wind speed, solar radiation) and flow values were allowed to change. However, climate change may also induce changes in land cover. Therefore, it may be appropriate to incorporate the effects of such changes in the model.

An uncertainty in the modeling of future water quality in the Spokane River came from the fact that the boundary conditions for future climate scenario simulation were kept unchanged. While modeling Lake Coeur D'Alene would be a significant undertaking, the impact on water temperature and other upstream boundary conditions at the Idaho-Washington Stateline would likely improve predictions in this study reach. The possibility that future climate change and population growth might alter parameter values was not taken into consideration in this study. Another important source of uncertainty was the assumptions made in order to generate future increases in point source discharges. The assumptions on population growth also had their own significant error bars. Although the calibrated model fitted the observed data well in this study, it is important to address the uncertainties in the model performance when the model is used in predictive mode outside the range of data used during model calibration. Nevertheless, the aforementioned assumptions and limitations need not undermine the findings of this study. In spite of the important gaps in data and uncertainties, models provide the only way to evaluate and understand the impacts of climate change on water quality. Results from this study, assessing the relative impacts of climate changes and population growth on Spokane River-Long Lake water quality, can be invaluable to water managers, even with the scope of uncertainties in modeling.

APPENDIX A

SUPPORTING MATERIALS FOR CHAPTER 3

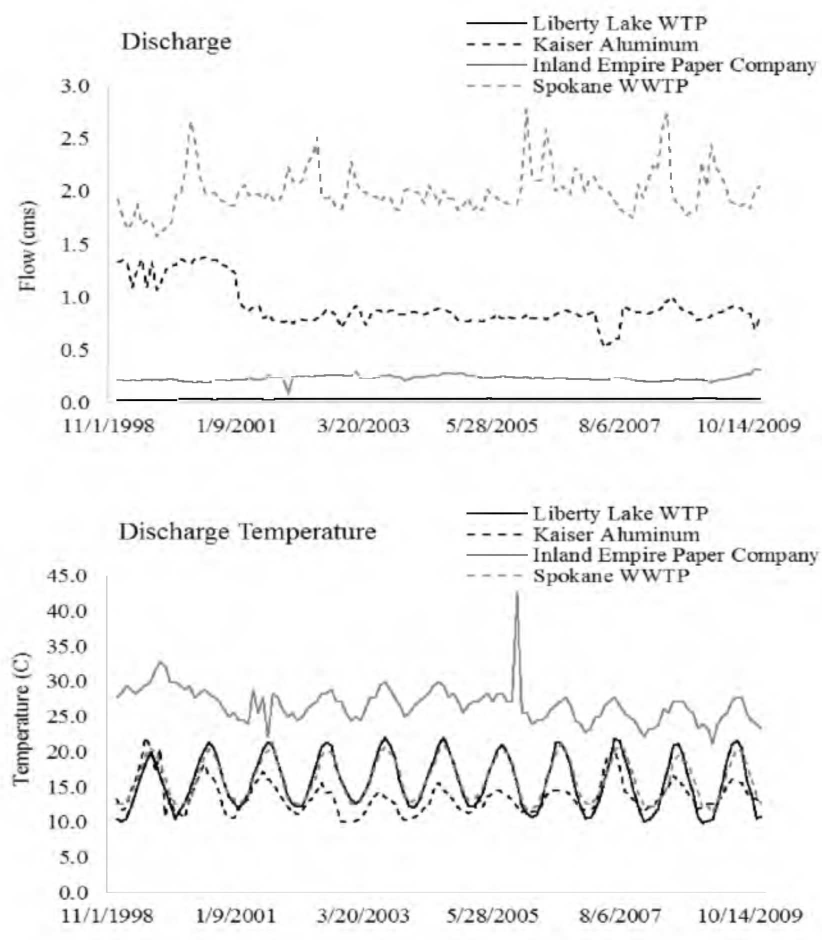


Figure A.1 Discharge and Discharge Temperatures at Point Dischargers

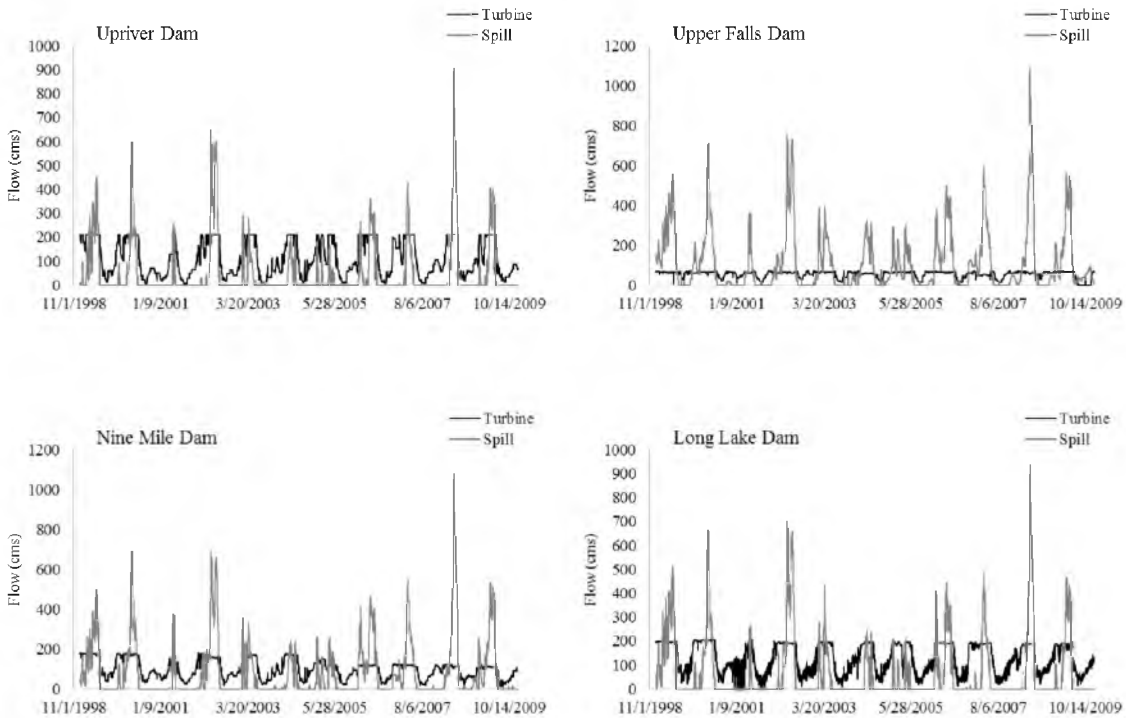


Figure A.2 Spill-Turbine Flow at Upriver, Upper Falls, Nine Mile, and Long Lake Dam

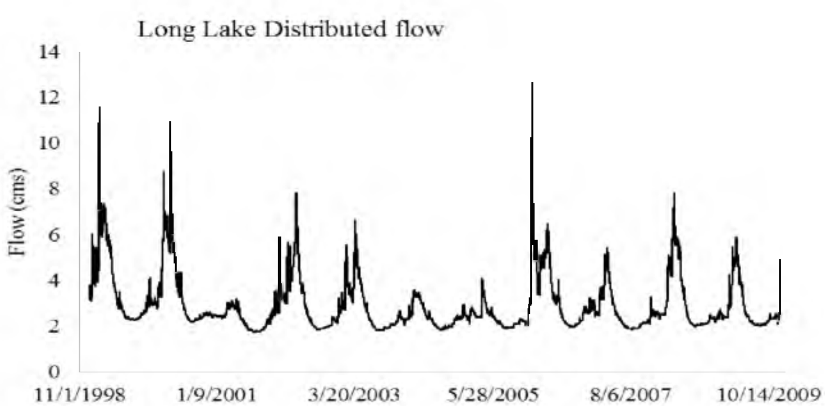


Figure A.3 Long Lake Distributed Flow

Table A.1 Comparison of Observed Meteorological Data

Variable	**Spokane International Airport**		**Spokane Felts Field**	
	Average	Range	Average	Range
Air temperature (°C)	8.8	-28.3 – 38.9	9.5	-28.3 – 38.9
Dew Point (°C)	1.0	-33.9 – 18.9	3.1	-28.0 – 27.2
Wind Speed (m/s)	4.0	0 – 22.6	2.3	0 – 19.0
Wind Direction (rad)	2.5	0 – 6.3	1.7	0 – 6.3
Cloud Cover	3.91	0 - 10	3.92	0 – 10

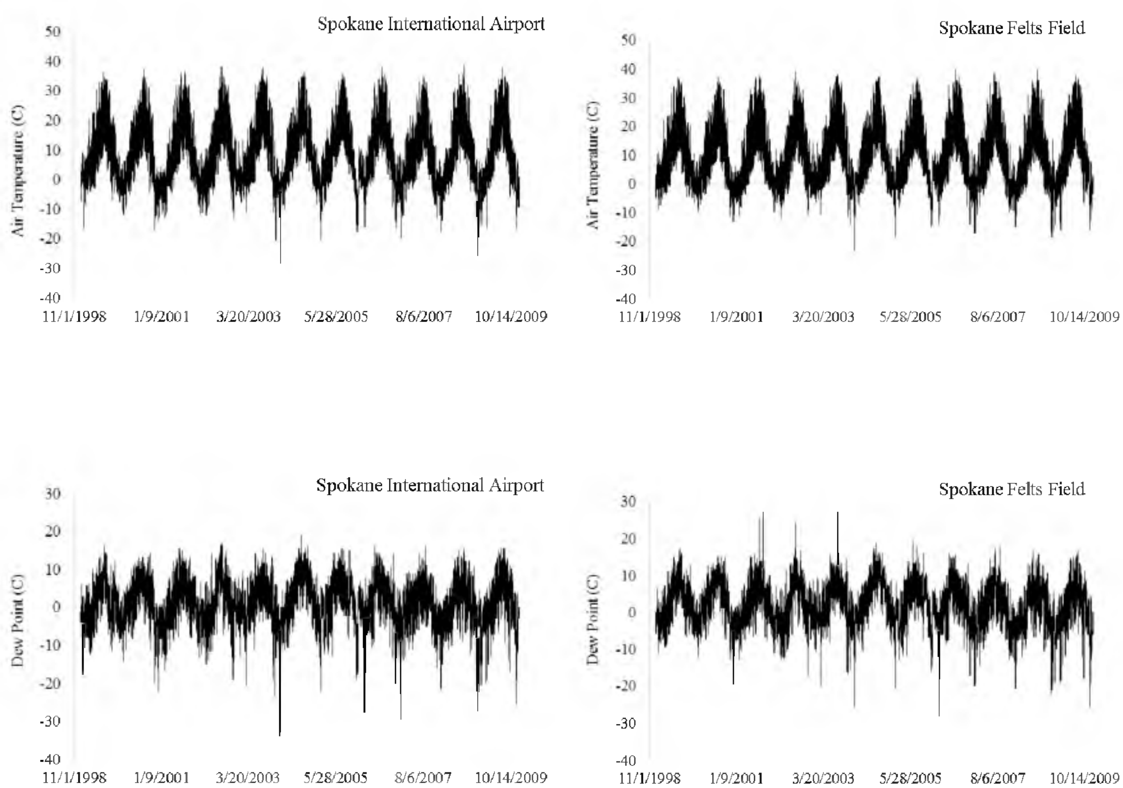


Figure A.4 Meteorological Data at Spokane International Airport, Spokane Felts Field

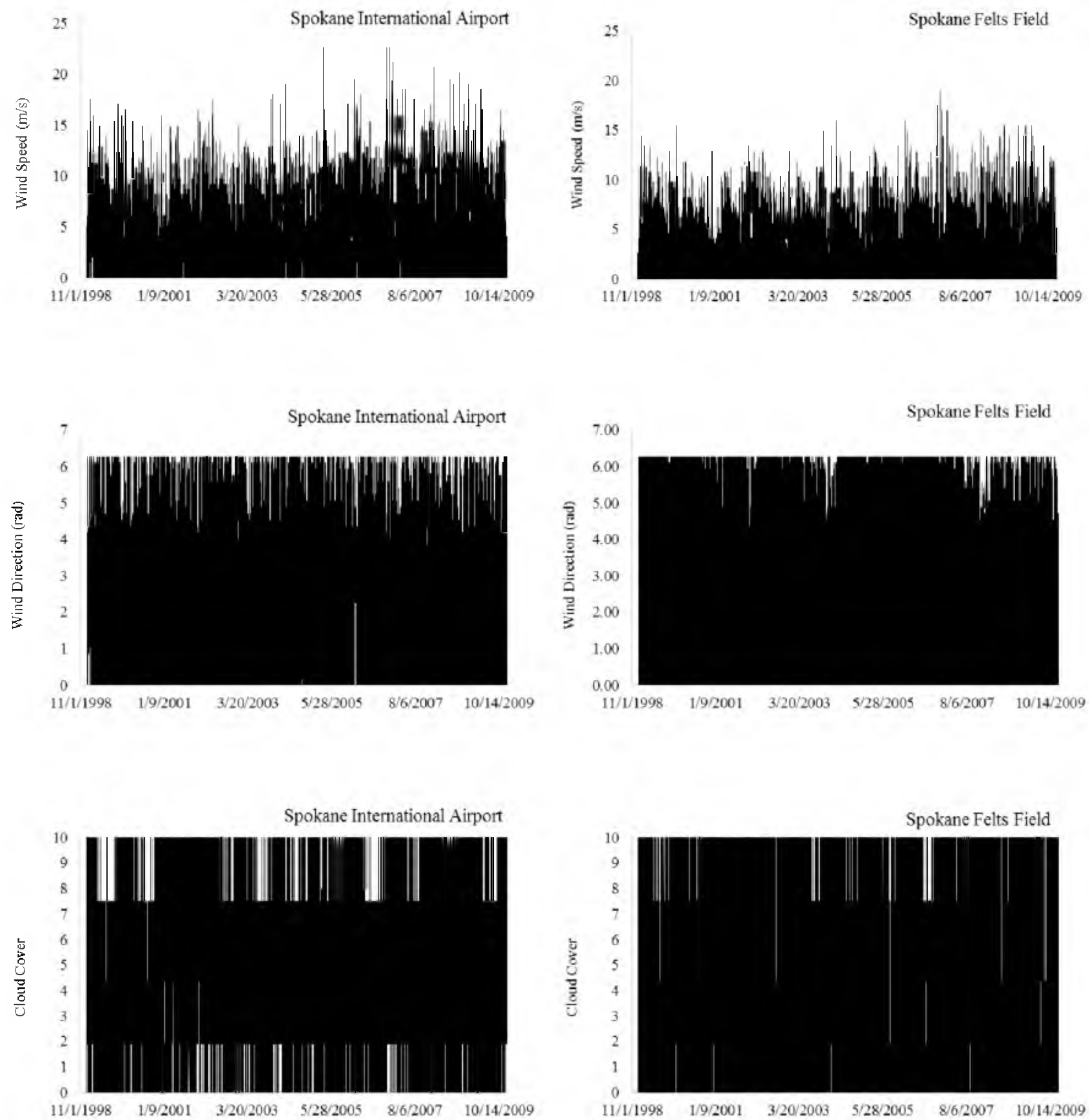


Figure A.4 Continued

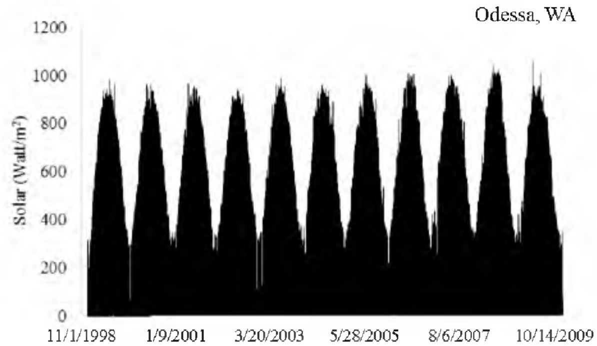


Figure A.5 Solar Radiation (W/m^2) at Odessa, WA

Table A.2 Comparison of RCP4.5 and RCP8.5 Projected Meteorological Data at Spokane International Airport and Spokane Felts Field

Meteorological Variable	**RCP4.5**		**RCP8.5**	
	Spokane International Airport			
	Mean	Std. Dev.	Mean	Std. Dev.
Air Temperature (C)	10.9	8.6	10.4	9.8
Dew Point (C)	1.8	6.0	1.3	7.4
Wind Speed (m/s)	2.4	1.3	2.4	1.3
Solar Radiation (W/m^2)	165.5	104.0	164.9	103.1
	Spokane Felts Field			
	Mean	Std. Dev.	Mean	Std. Dev.
Air Temperature (C)	11.7	8.4	11.2	9.6
Dew Point (C)	2.3	5.6	1.9	7.0
Wind Speed (m/s)	2.4	1.3	2.4	1.2
Solar Radiation (W/m^2)	159.5	100.9	158.6	100.2

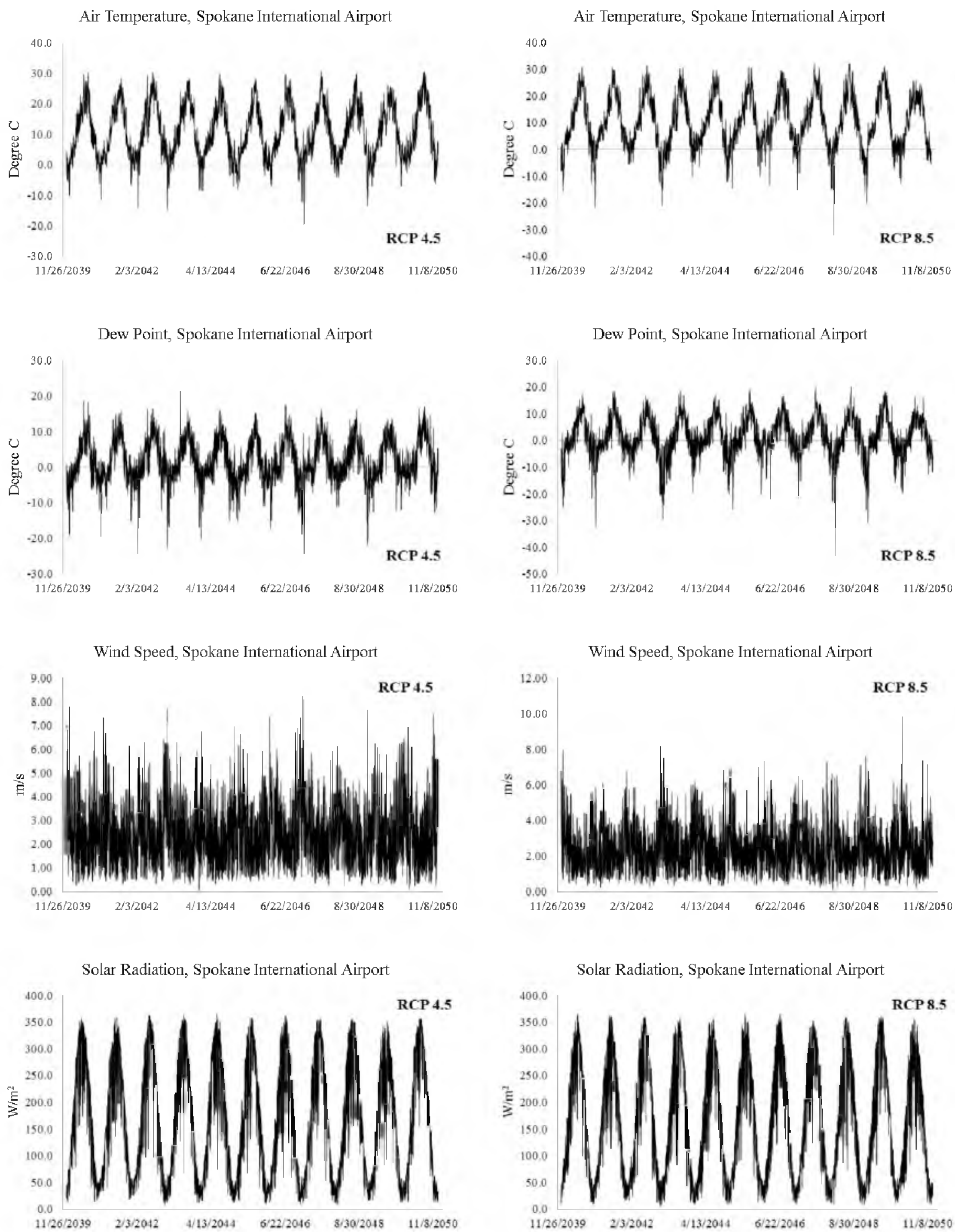


Figure A.6 Projected Meteorological Data at Spokane International Airport (CNRM-CM5 Model, RCP4.5 and RCP8.5 Scenarios)

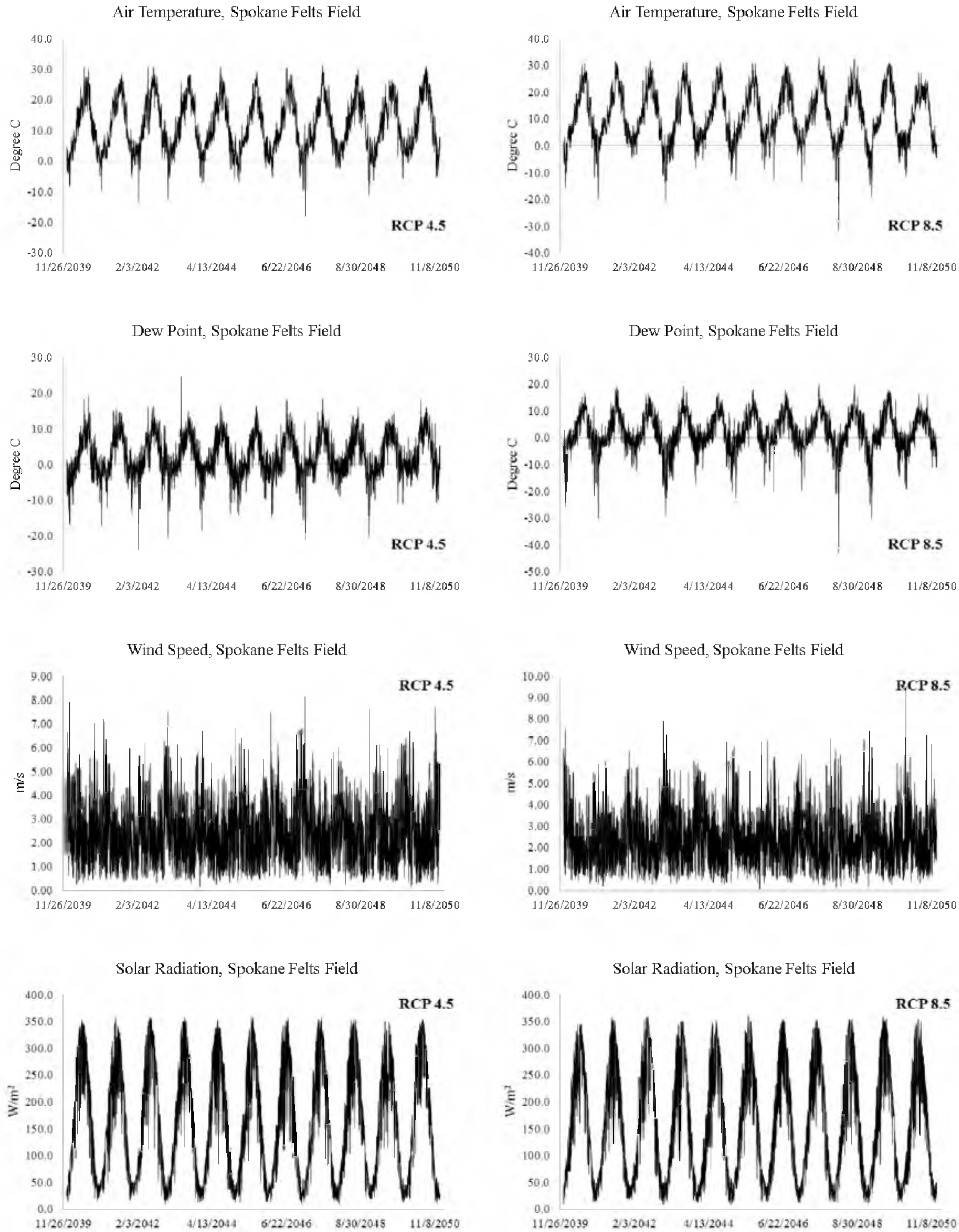


Figure A.7 Projected Meteorological Data at Spokane Felts Field (CNRM-CM5 Model, RCP4.5 and RCP8.5 Scenarios)

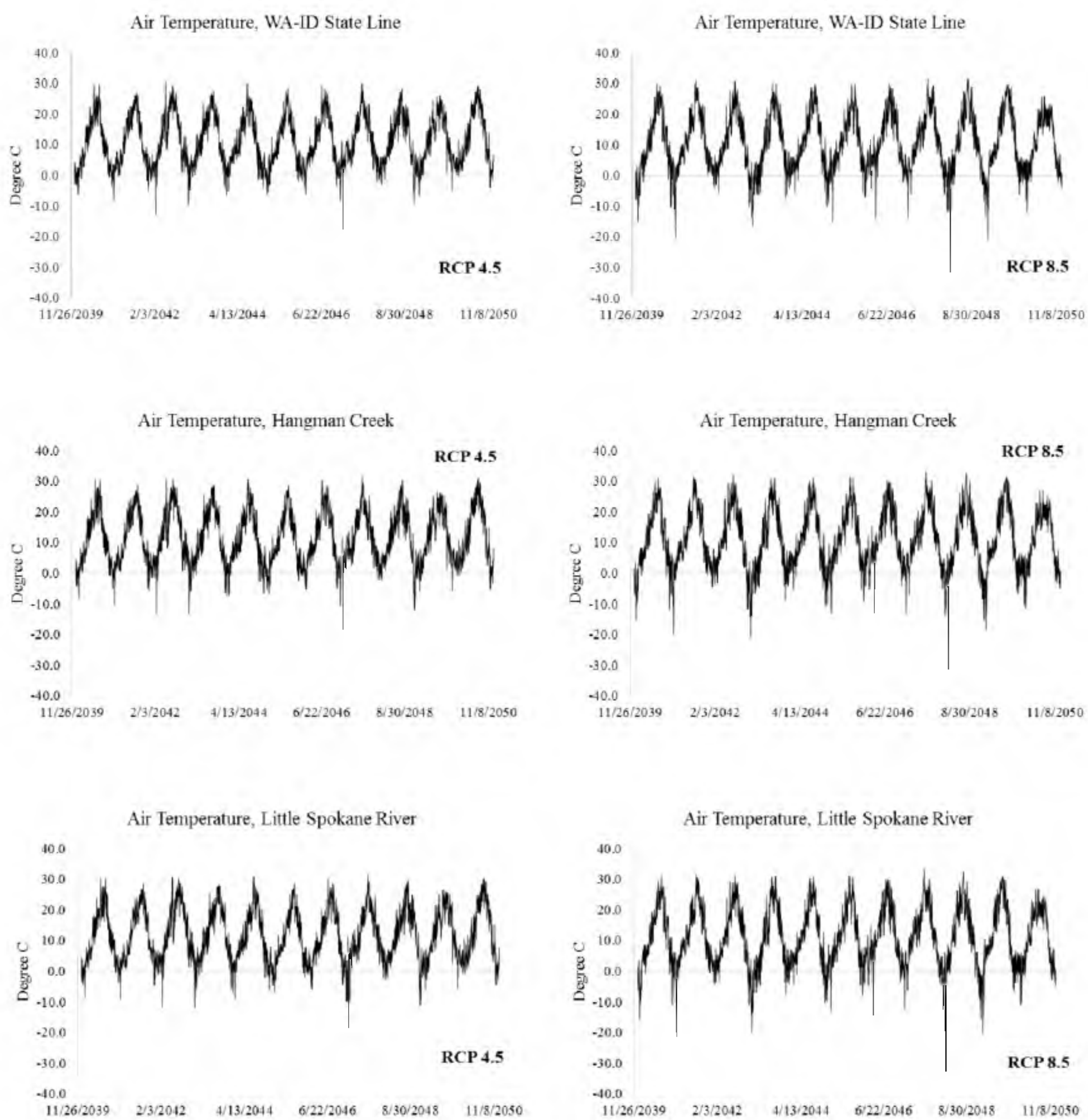


Figure A.8 Projected Air Temperature Data at WA-ID Stateline, Hangman Creek and Little Spokane River (RCP4.5 and RCP8.5 Scenarios)

Table A.3 Comparison of RCP4.5 and RCP8.5 Air Temperature Data Projections

Meteorological Variable	**RCP4.5**		**RCP8.5**	
	Mean (C)	Std. Dev. (C)	Mean (C)	Std. Dev. (C)
WA-ID Stateline	11.2	8.1	10.8	9.3
Hangman Creek	11.7	8.5	11.2	9.7
Little Spokane River	11.8	8.4	11.3	9.6

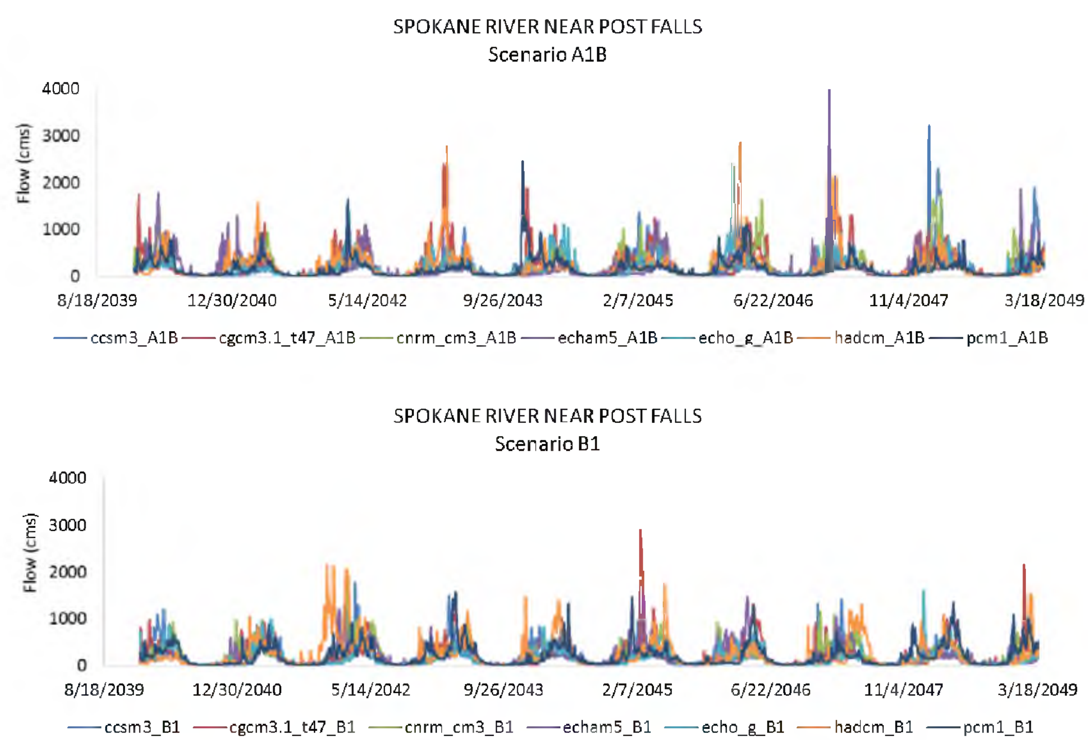


Figure A.9 Projected Streamflows at Stateline, Hangman Creek, Little Spokane River

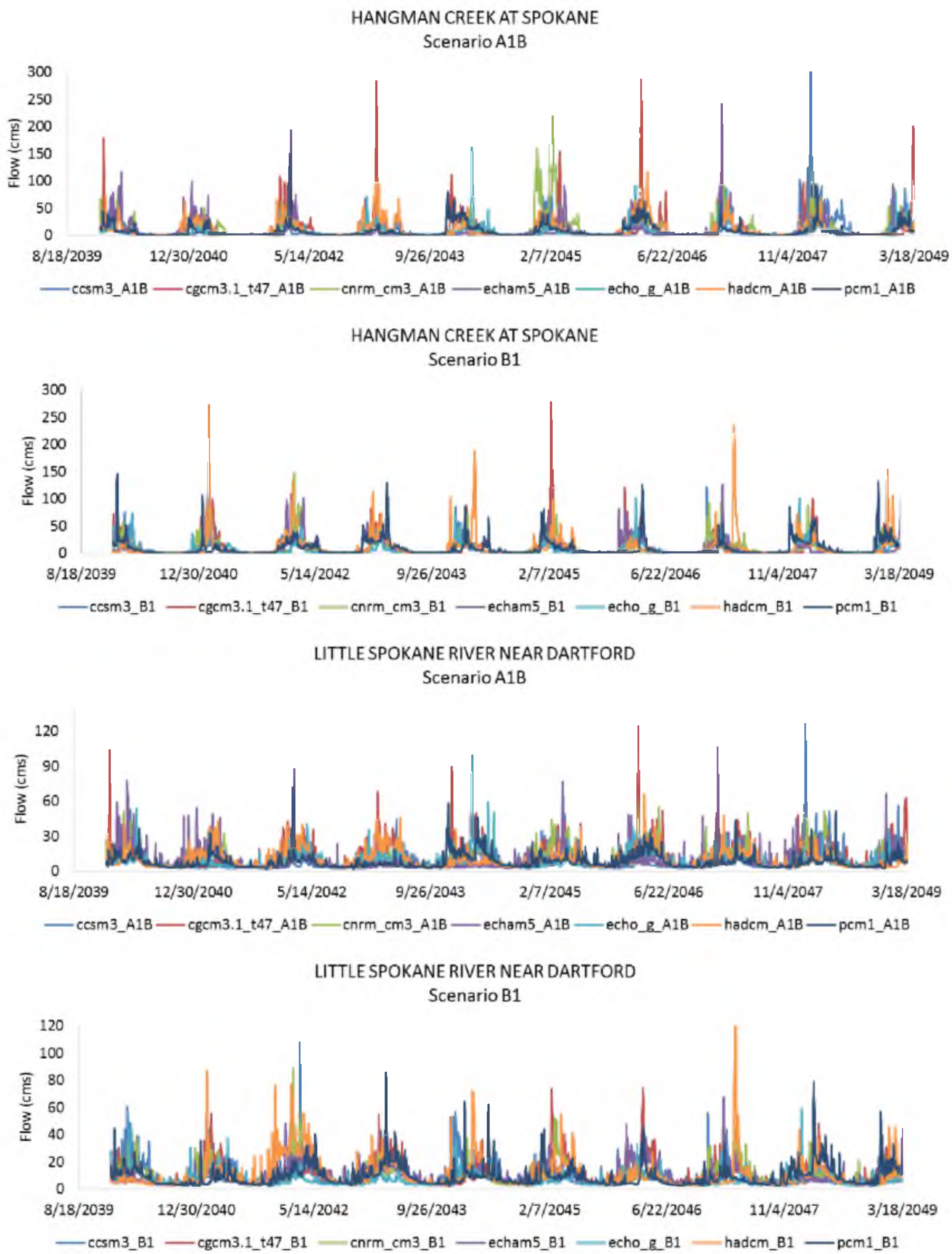


Figure A.9 Continued

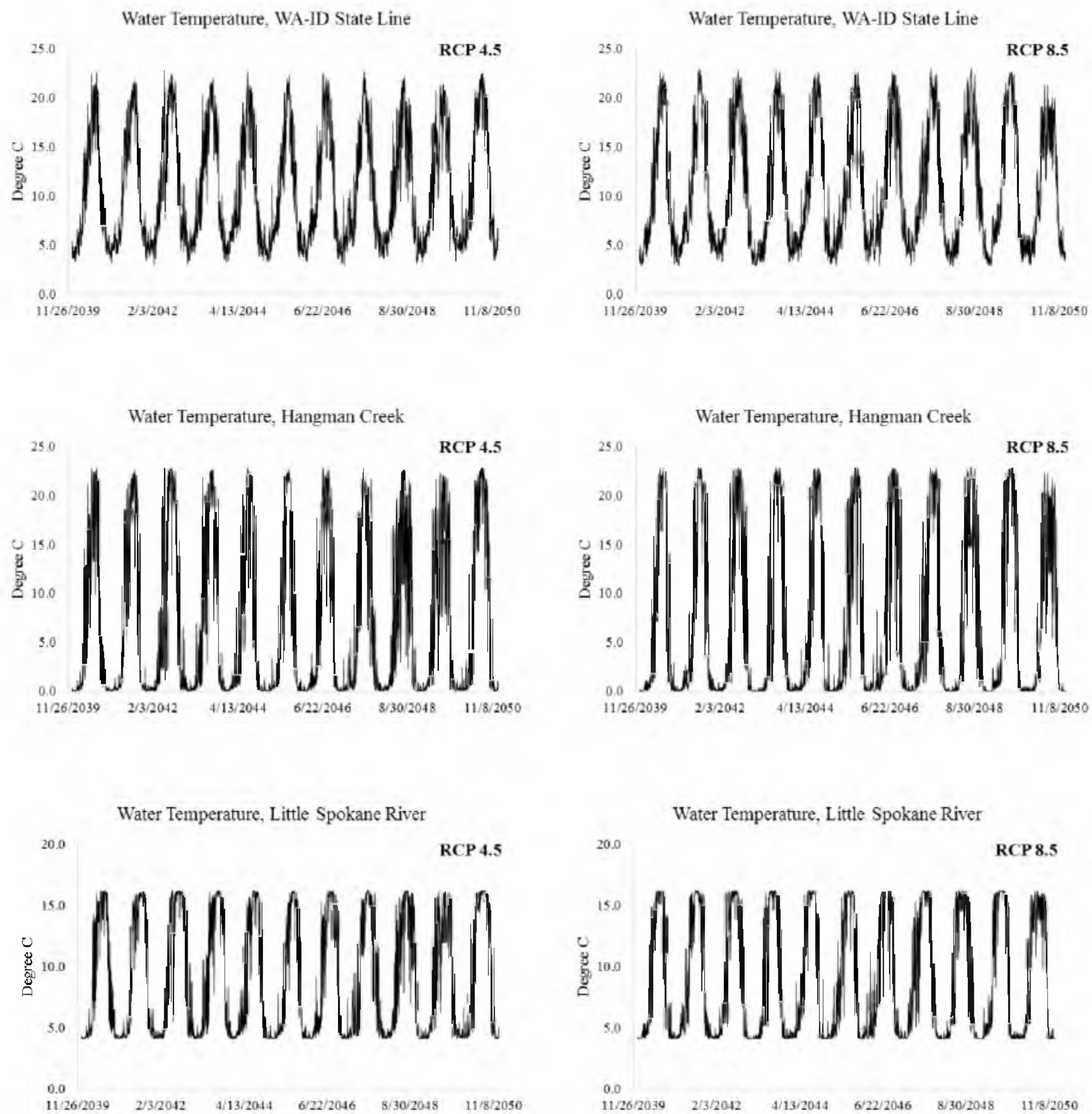


Figure A.10 Projected Water Temperature Data at WA-ID Stateline, Hangman Creek and Little Spokane River (RCP4.5 and RCP8.5 Scenarios)

Table A.4 Comparison of Projected Water Temperature Data at WA-ID Stateline, Hangman Creek and Little Spokane River

Meteorological Variable	**RCP4.5**		**RCP8.5**	
	Mean (C)	Std. Dev. (C)	Mean (C)	Std. Dev. (C)
WA-ID Stateline	10.9	5.8	10.8	6.1
Hangman Creek	7.2	8.1	7.2	8.4
Little Spokane River	9.1	4.8	9.0	4.9

APPENDIX B

SUPPORTING MATERIALS FOR CHAPTER 4

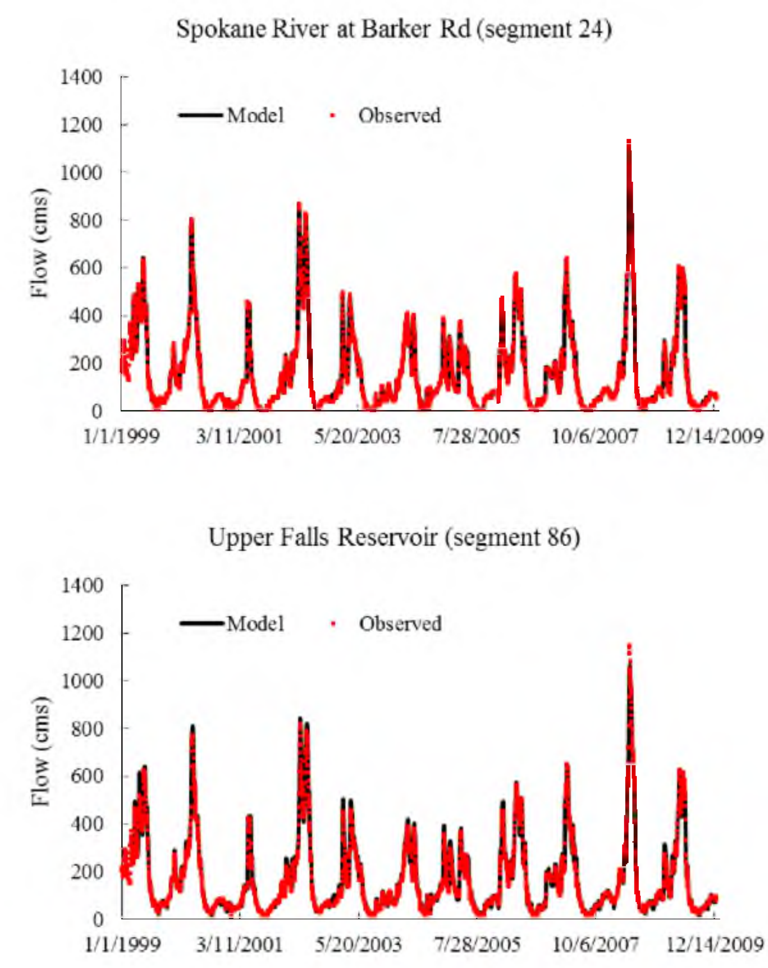
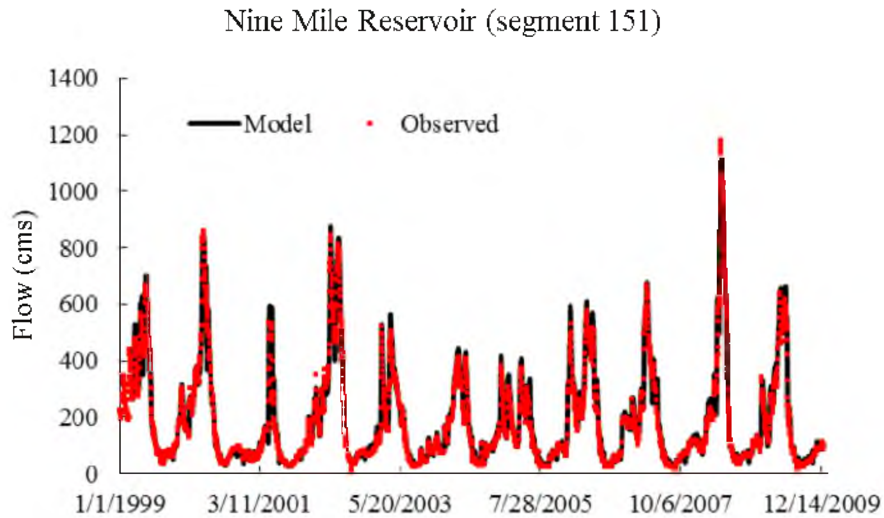


Figure B.1 Flow Prediction Comparison with Observed Data



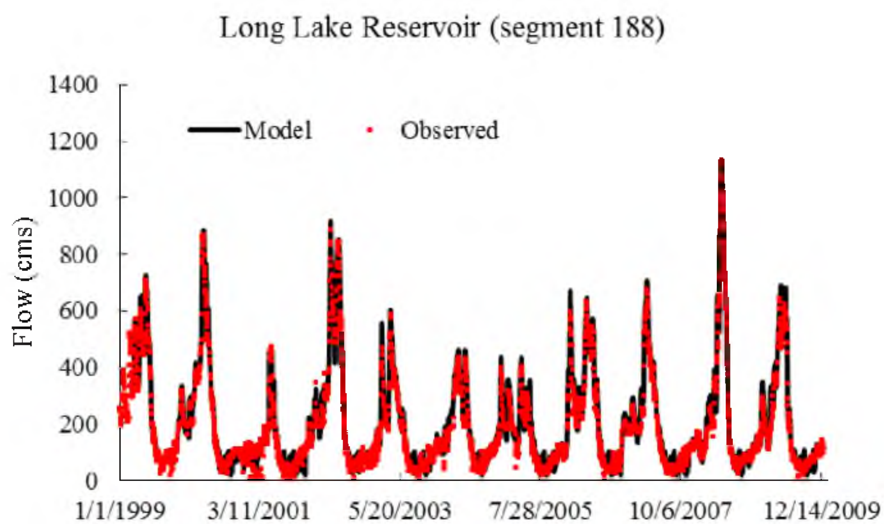


Figure B.1 Continued

Table B.1 Long Lake and Spokane River Temperature Profiles Sites for 2000-2001

Site ID	Description	Segment No.	RM
SPK82.5	Spokane River above Upriver Dam: 2.7 miles upstream of dam	57	80.2
SPK81.6	Spokane River above Upriver Dam: 1.8 miles upstream of dam	60	81.6
SPK81.0	Spokane River above Upriver Dam: 1.2 miles upstream of dam	62	81.0
SPK79.8	Spokane River above Upriver Dam: 0.4 miles upstream of dam	64	79.8
SPK62.0	Spokane River above Nine mile Dam: 3.8 miles upstream of dam	135	62.0
SPK61.4	Spokane River above Nine mile Dam: 3.3 miles upstream of dam	139	61.4
SPK60.9	Spokane River above Nine mile Dam: 2.8 miles upstream of dam	141	60.9
SPK60.2	Spokane River above Nine mile Dam: 2.1 miles upstream of dam	143	60.2
SPK58.9	Spokane River above Nine mile Dam: 0.8 miles upstream of dam	147	58.9
SPK58.3	Spokane River above Nine mile Dam: 0.2 miles upstream of dam	150	58.3
LL5	Long Lake @ Station 5	157	54.2
LL4	Long Lake @ Station 4	161	51.5
LL3	Long Lake @ Station 3	168	46.4
LL2	Long Lake @ Station 2	174	42.1
LL1	Long Lake @ Station 1	180	37.6
LL0	Long Lake @ Station 0	187	32.7

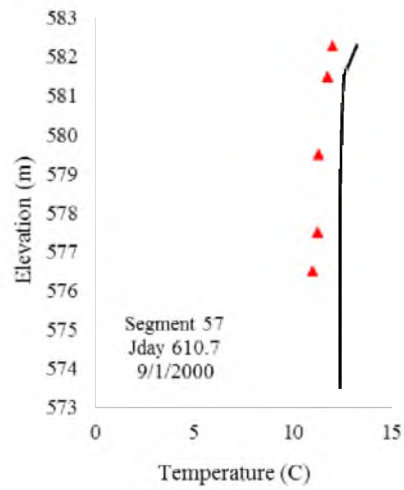


Figure B.2 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 2.7 Miles Upstream of Upriver Dam (Segment 57)

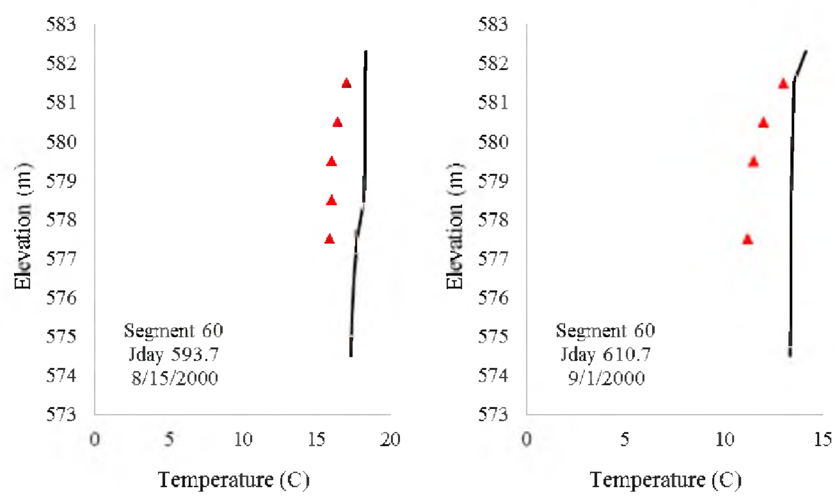


Figure B.3 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 1.8 Miles Upstream of Upriver Dam (Segment 60)

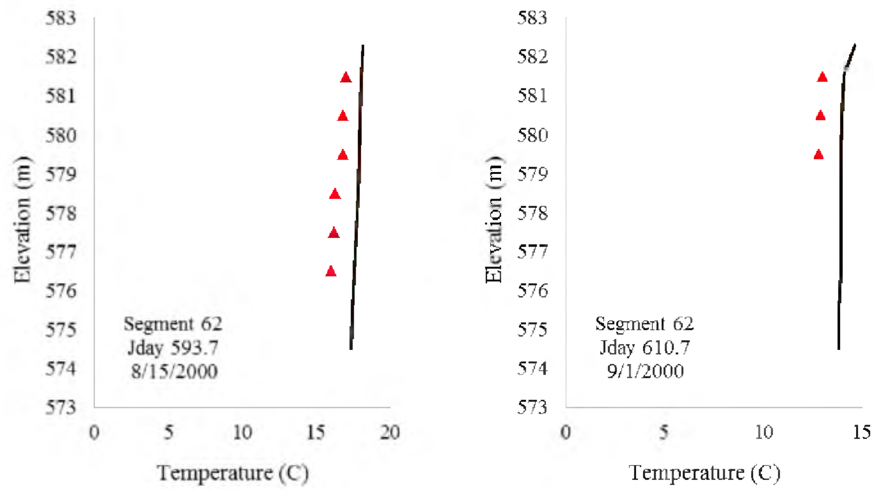


Figure B.4 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 1.2 Miles Upstream of Upriver Dam (Segment 62)

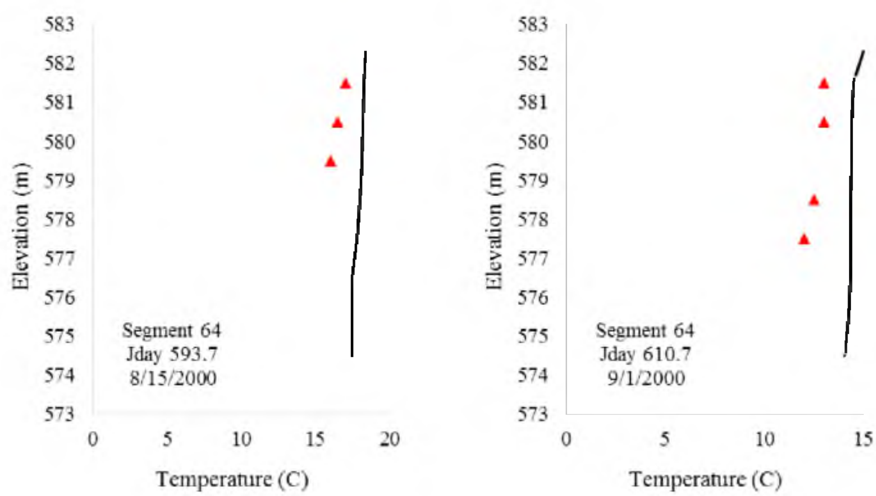


Figure B.5 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 0.4 Miles Upstream of Upriver Dam (Segment 64)

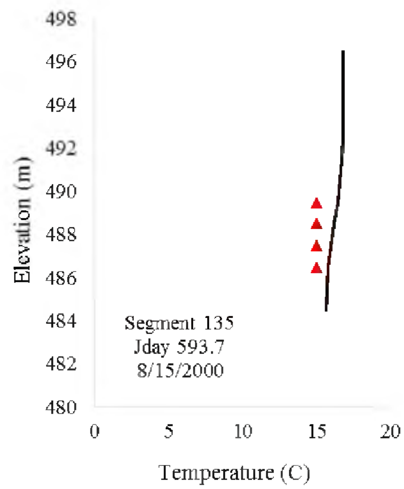


Figure B.6 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 3.8 Miles Upstream of Nine Mile Dam (Segment 135)

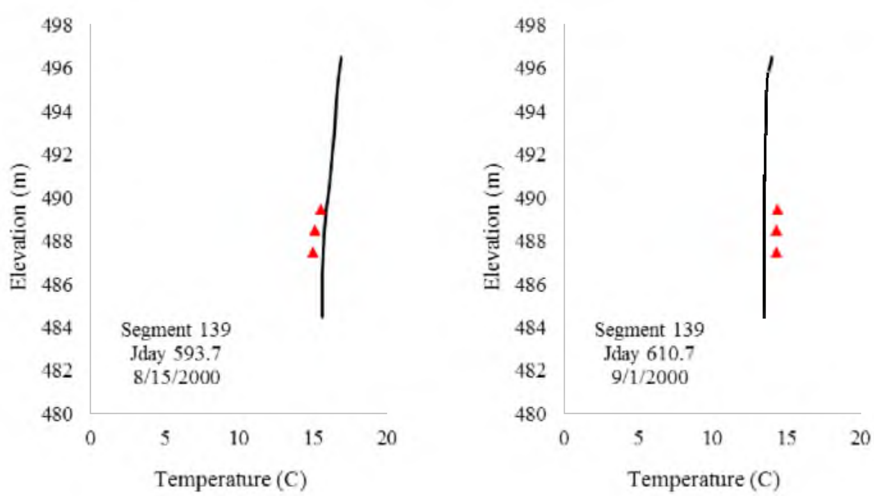


Figure B.7 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 3.3 Miles Upstream of Nine Mile Dam (Segment 139)

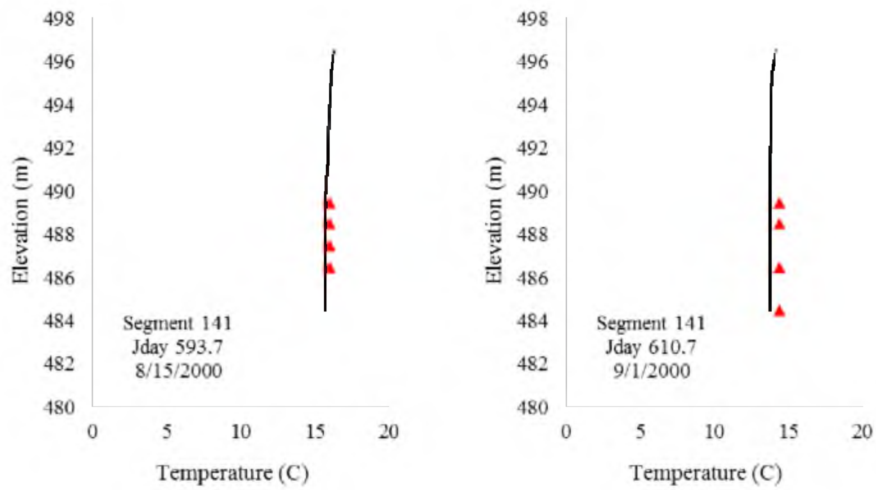


Figure B.8 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 2.8 Miles Upstream of Nine Mile Dam (Segment 141)

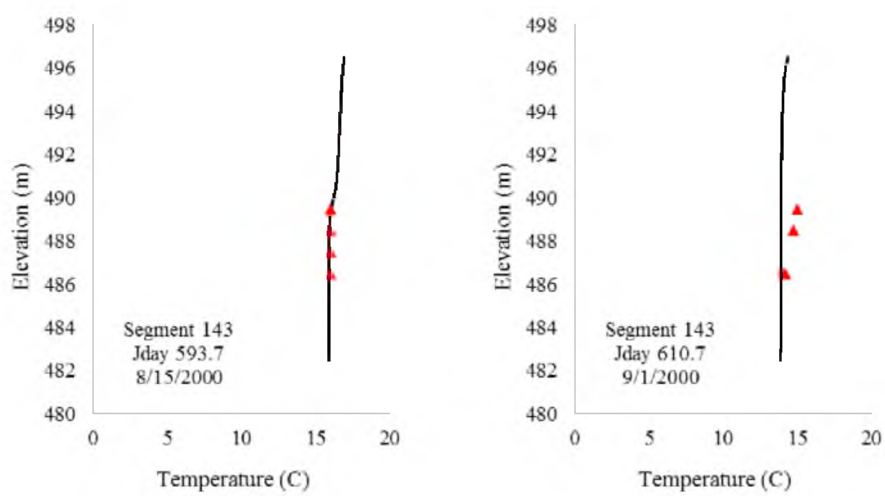


Figure B.9 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 2.1 Miles Upstream of Nine Mile Dam (Segment 143)

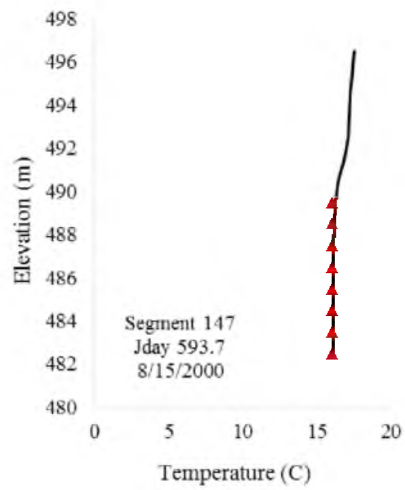


Figure B.10 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 0.8 Miles Upstream of Nine Mile Dam (Segment 147)

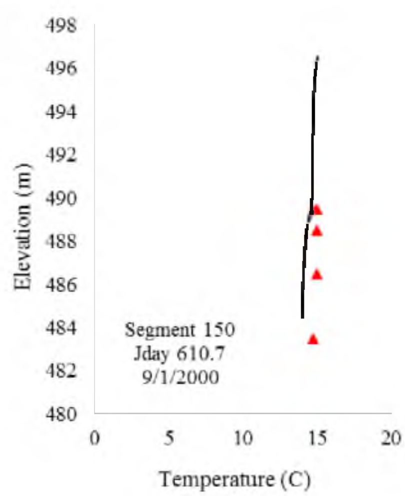


Figure B.11 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 0.2 Miles Upstream of Nine Mile Dam (Segment 150)

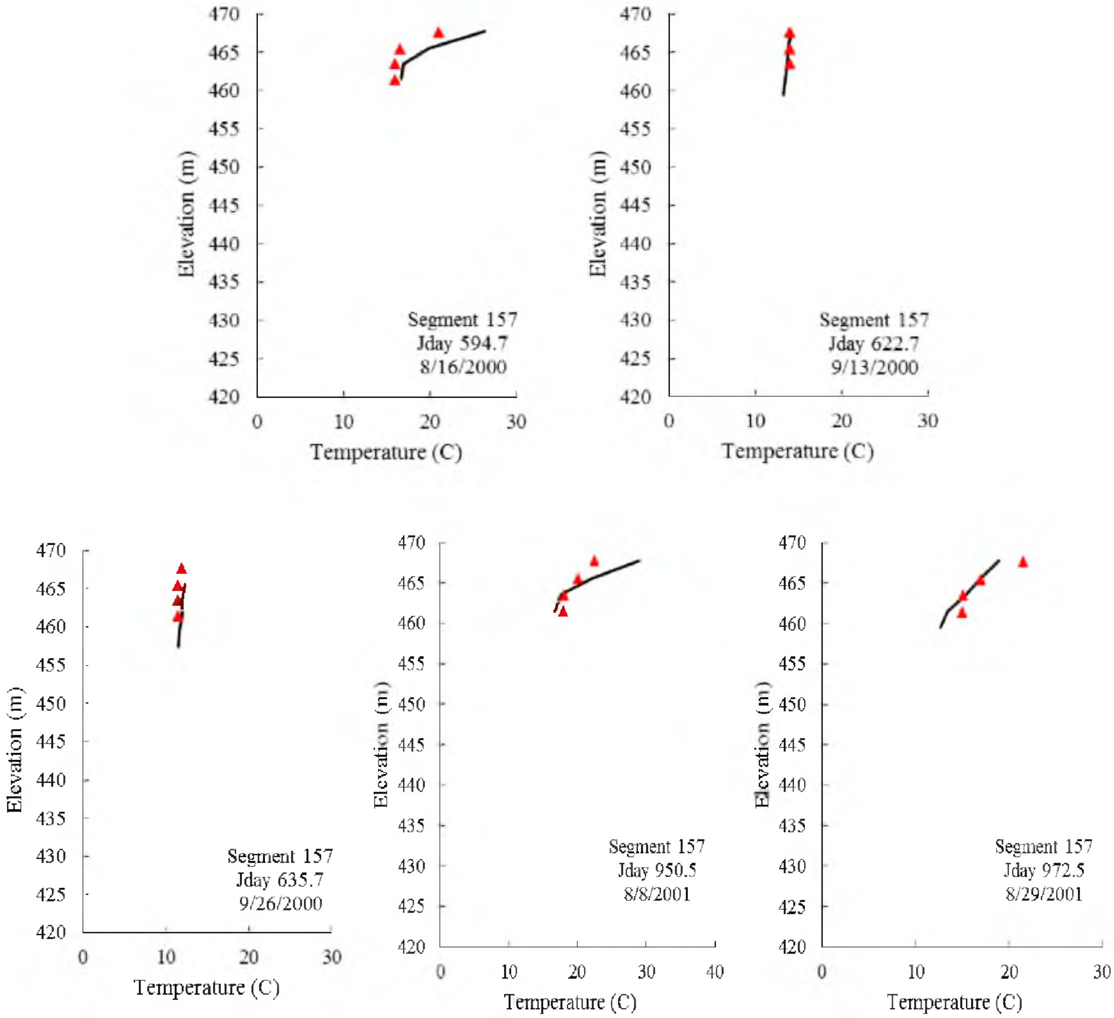


Figure B.12 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 5 (Segment 157)

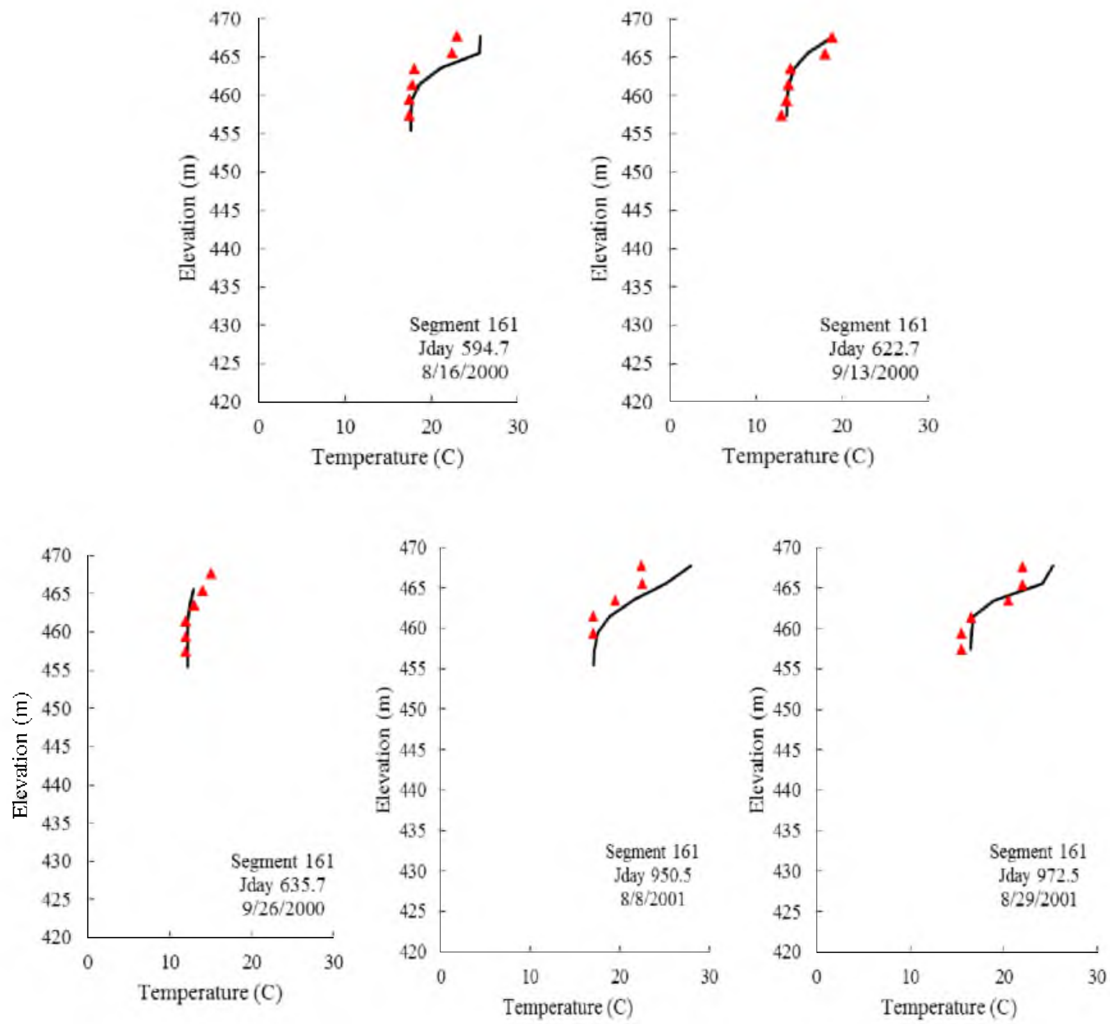


Figure B.13 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 4 (Segment 161)

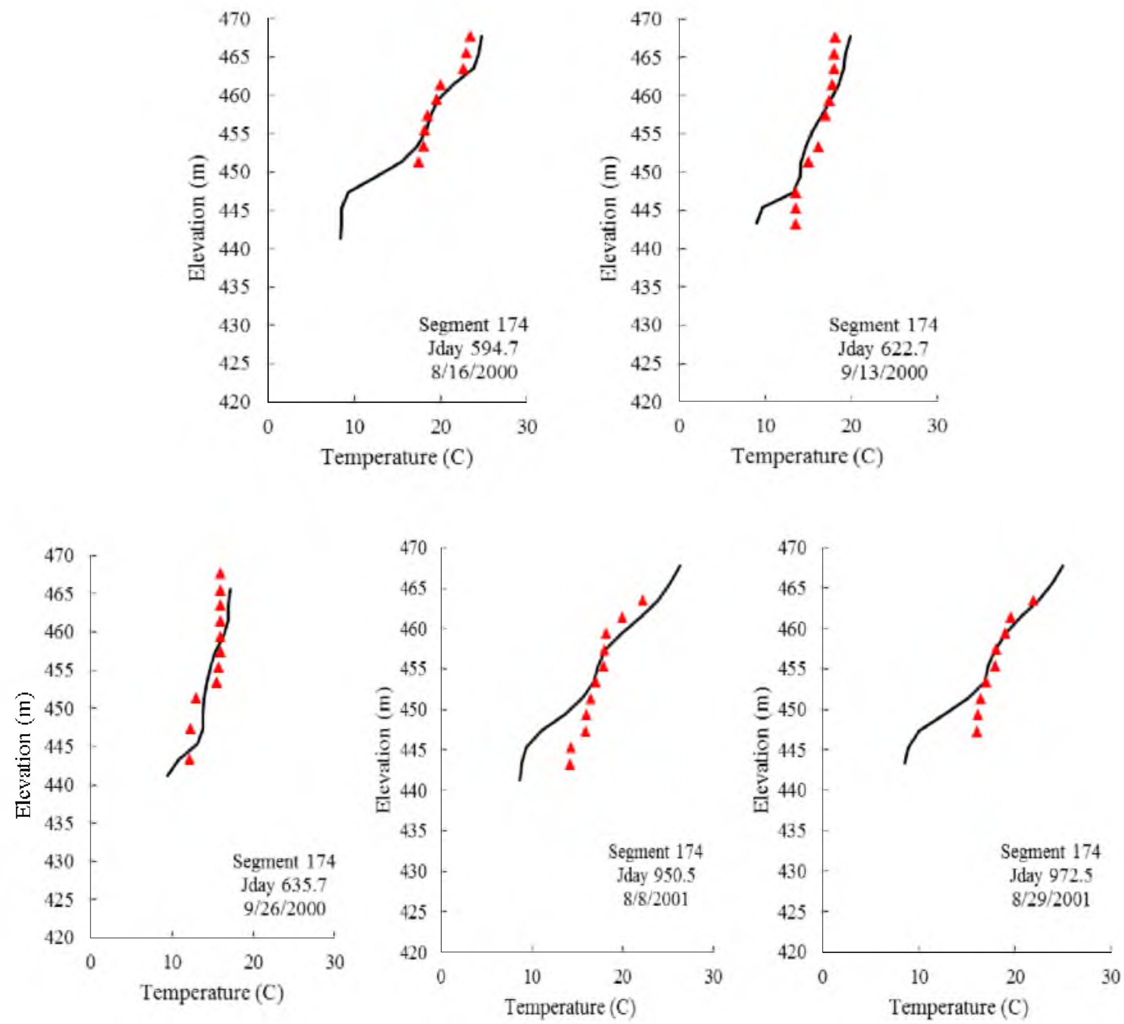


Figure B.14 Comparison of Model Predicted Vertical Temperature Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 2 (Segment 174)

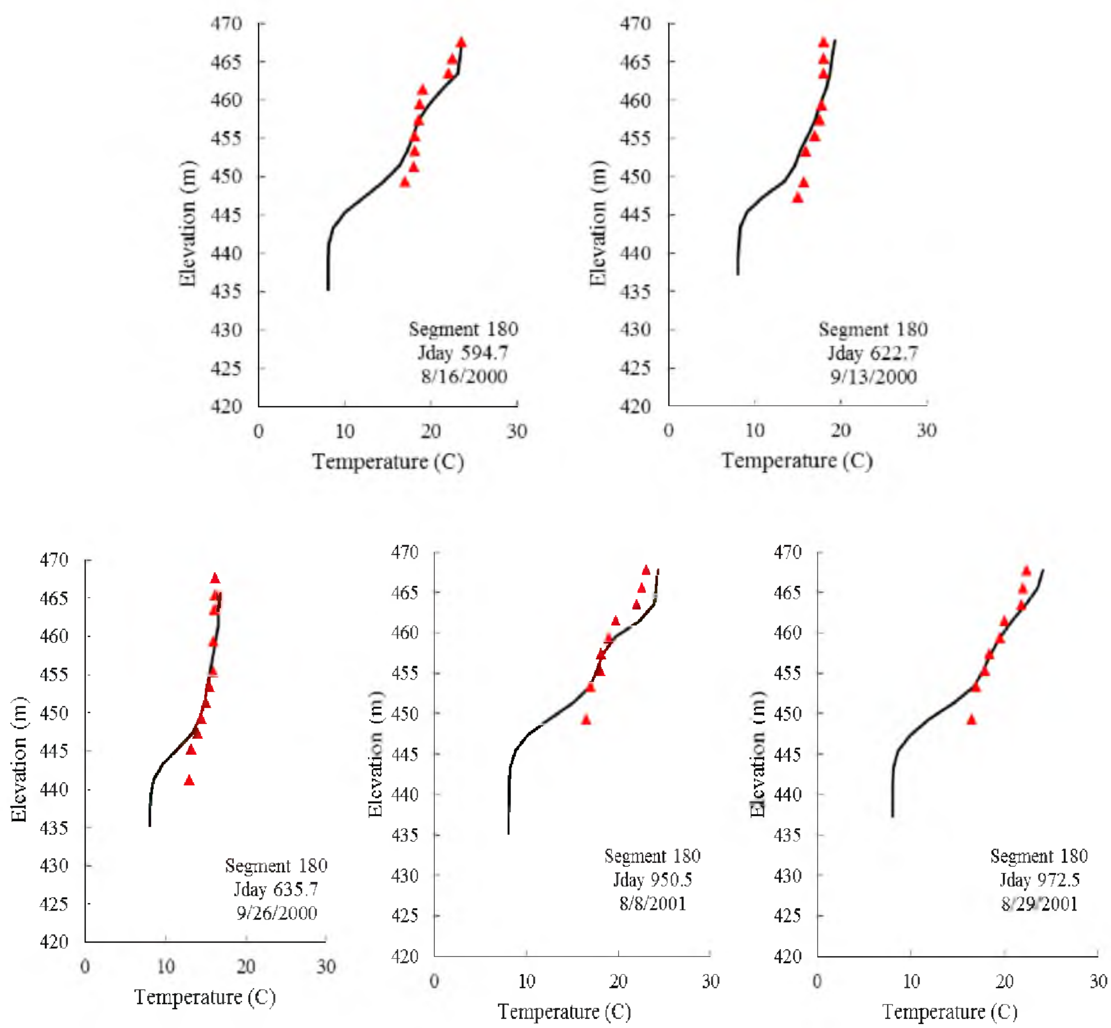


Figure B.15 Comparison of Model Predicted Vertical Temperature Profile (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 1 (Segment 180)

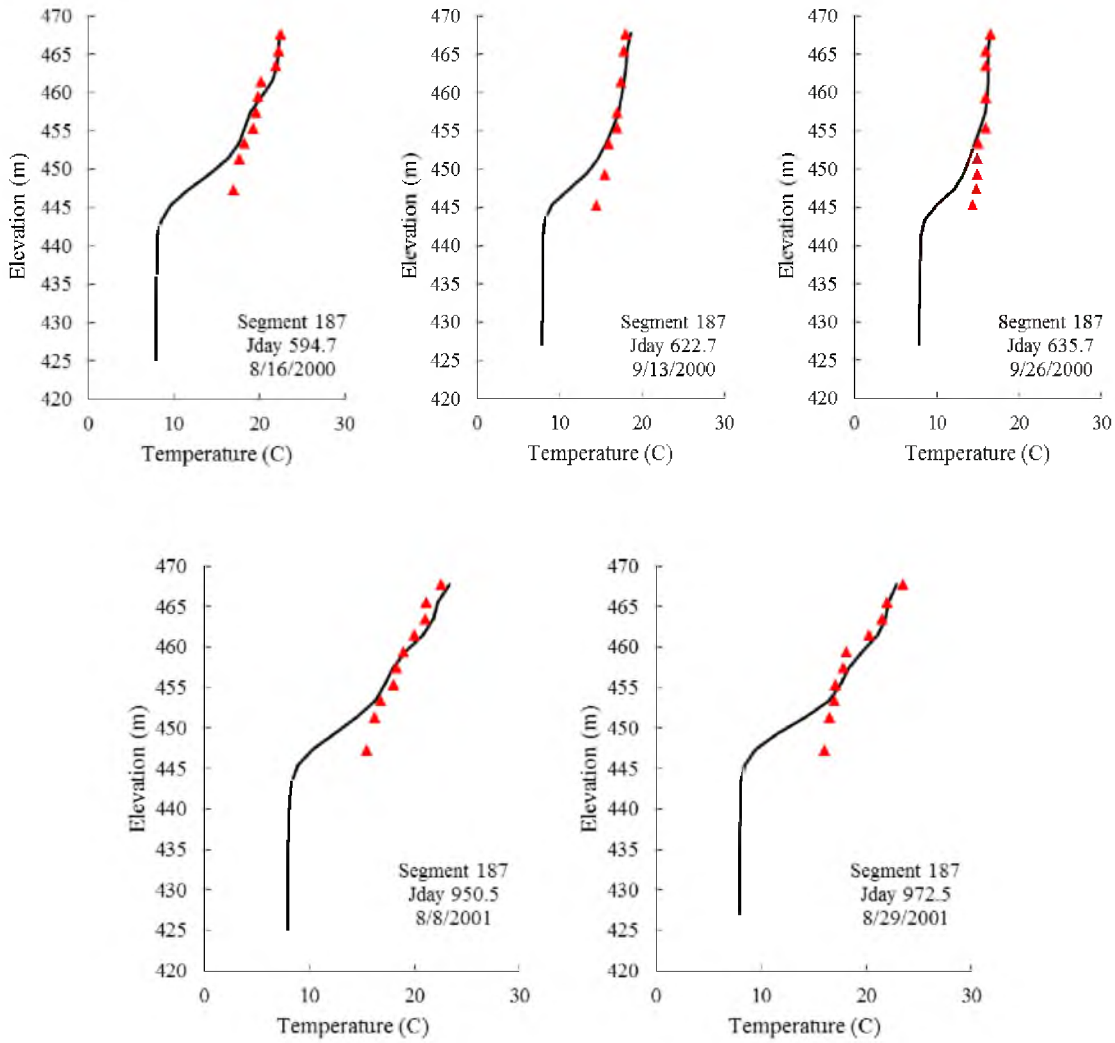


Figure B.16 Comparison of Model Predicted Vertical Temperature Profile (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 0 (Segment 187)

Table B.2 Profile Error Statistics

Location	N, # of data profile comparisons	Error statistics (^oC)	
		AME	**RMS**
SPK82.5, Segment 57	1	1.14	1.16
SPK81.6, Segment 60	2	1.72	1.80
SPK81.0, Segment 62	2	1.28	1.29
SPK79.8, Segment 64	2	1.75	1.79
SPK62.0, Segment 135	1	1.08	1.11
SPK61.4, Segment 139	2	0.71	0.73
SPK60.9, Segment 141	2	0.46	0.49
SPK60.2, Segment 143	2	0.36	0.53
SPK58.9, Segment 147	1	0.14	0.14
SPK58.3, Segment 150	1	0.66	0.70
LL5, Segment 157	5	0.97	1.35
LL4, Segment 161	5	1.34	1.87
LL3, Segment 168	5	0.88	1.19
LL2, Segment 174	5	1.02	1.33
LL1, Segment 180	5	1.07	1.53
LL0, Segment 187	5	0.96	1.56

Table B.3 Water Quality Data Time Series Sites for 1999-2009

Description	Segment No.	RM
Baker Road	24	90.3
Sullivan Road	36	87.8
Plante's Ferry Park	48	8B.7
Above Upriver Dam	67	79.7
Green Street Bridge	73	78.0
Sandifer Bridge	97	72.6
Fort Wright Bridge	106	69.8
Above Spokane WWTP	114	67.6
Riverside State Park	119	66.0
Seven Mile Bridge	135	62.0
Nine Mile Dam	151	58.1
Long Lake Dam	188	33.5

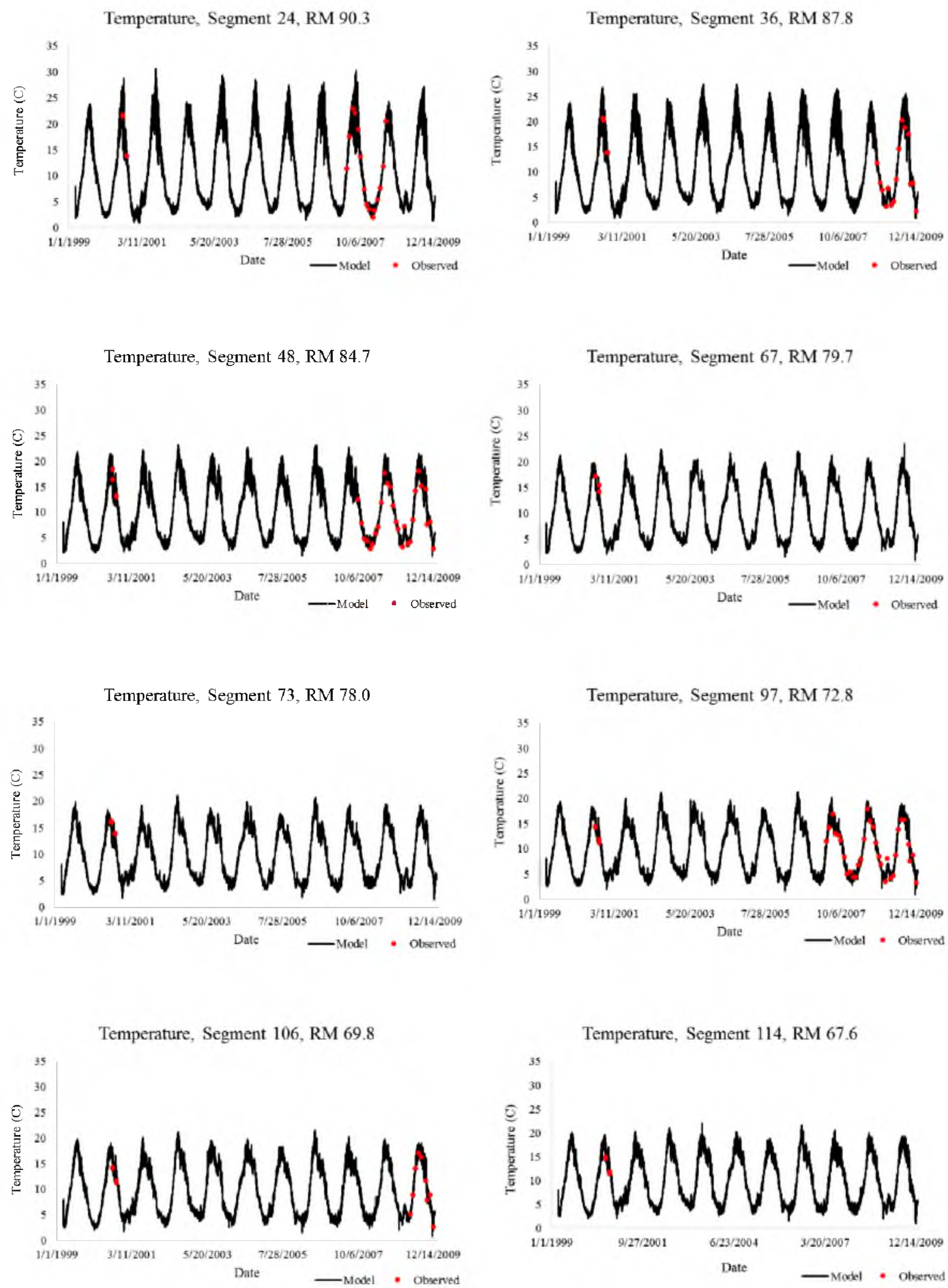


Figure B.17 Time Series Comparisons of Temperature Data

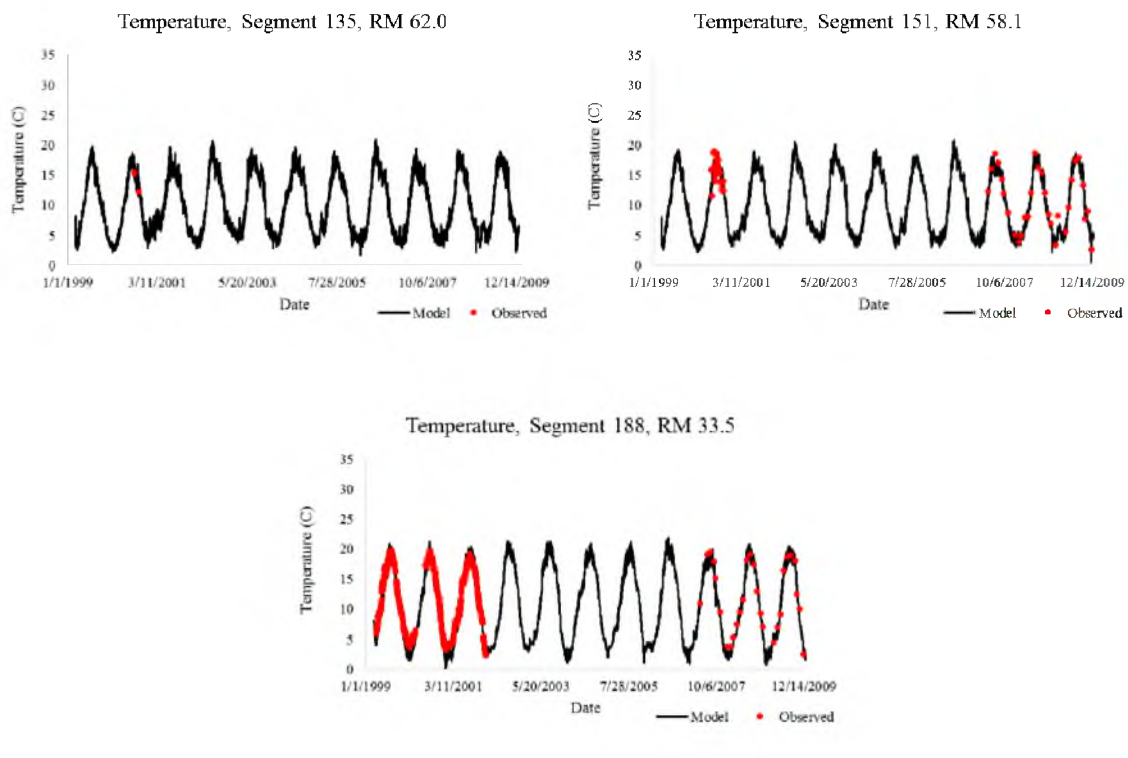


Figure B.17 Continued

Table B.4 Temperature Time Series Error Statistics

Location	RM	N, # of data	Error statistics (°C)	
			AME	RMS
Baker Road	90.3	19	0.83	1.26
Sullivan Road	87.8	18	0.87	1.44
Plante's Ferry Park	8B.7	27	1.13	3.01
Above Upriver Dam	79.7	4	0.63	0.79
Green Street Bridge	78.0	4	0.40	0.41
Sandifer Bridge	72.6	32	0.72	0.98
Fort Wright Bridge	69.8	13	1.03	1.32
Above Spokane WWTP	67.6	4	1.71	1.75
Riverside State Park	66.0	136	1.00	1.96
Seven Mile Bridge	62.0	4	0.91	0.92
Nine Mile Dam	58.1	56	1.05	1.42
Long Lake Dam	33.5	921	1.07	1.31

Table B.5 Temperature Time Series Statistics (^{o}C)

Location	RM	# of Data	Model Data		Observed Data	
			Mean	Std. Dev.	Mean	Std. Dev.
Baker Road	90.3	19	13.6	7.3	12.8	7.0
Sullivan Road	87.8	18	11.4	7.1	11.0	6.1
Plante's Ferry Park	84.7	27	9.8	4.9	9.3	4.7
Above Upriver Dam	79.7	4	16.1	1.7	16.0	1.3
Green Street Bridge	78.0	4	14.8	1.0	15.0	1.2
Sandifer Bridge	72.6	32	9.9	4.7	9.8	4.3
Fort Wright Bridge	69.8	13	12.0	4.8	11.1	4.1
Above Spokane WWTP	67.6	4	15.5	2.0	13.2	1.5
Riverside State Park	66.0	136	10.3	5.2	9.7	4.7
Seven Mile Bridge	62.0	4	15.3	1.9	13.8	1.6
Nine Mile Dam	58.1	56	12.5	4.8	13.0	4.4
Long Lake Dam	33.5	921	10.7	6.1	9.0	5.4

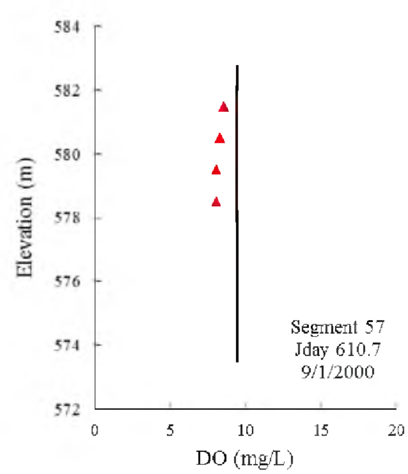


Figure B.18 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 2.7 Miles Upstream of Upriver Dam (Segment 57)

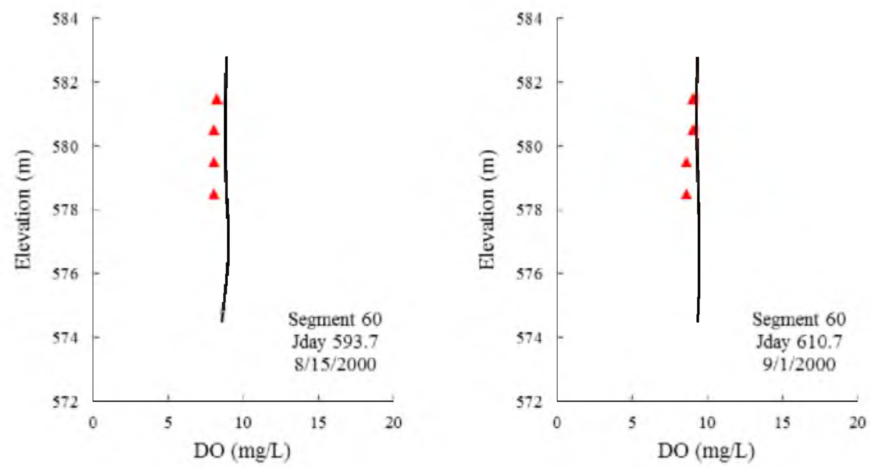


Figure B.19 Comparison of Model Predicted Vertical DO Profiles and 2000 Data for the Spokane River 1.8 Miles Upstream of Upriver Dam (Segment 60)

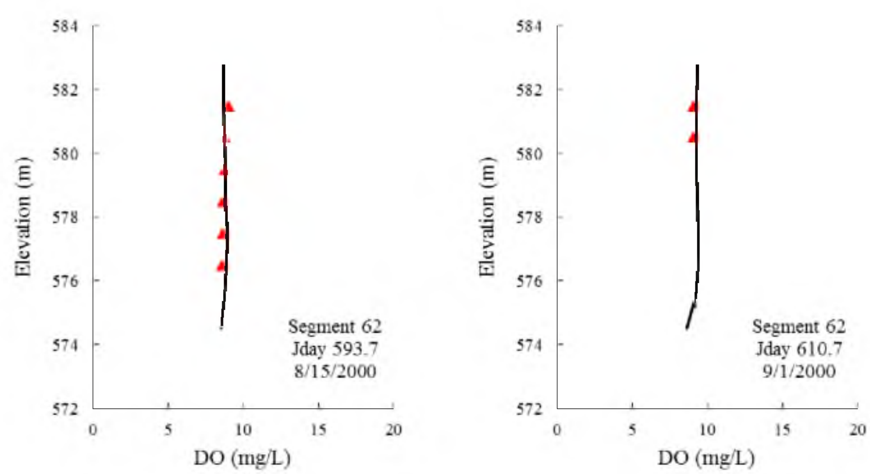


Figure B.20 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 1.2 Miles Upstream of Upriver Dam (Segment 62)

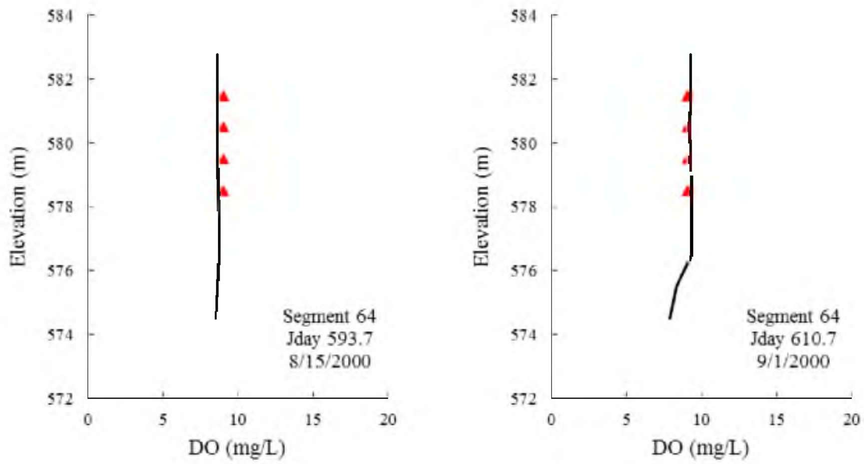


Figure B.21 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 0.4 Miles Upstream of Upriver Dam (Segment 64)

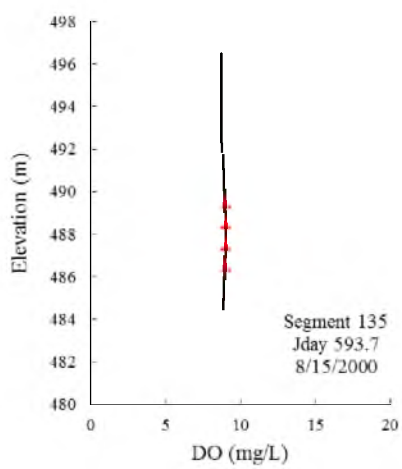


Figure B.22 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 3.8 Miles Upstream of Nine Mile Dam (Segment 135)

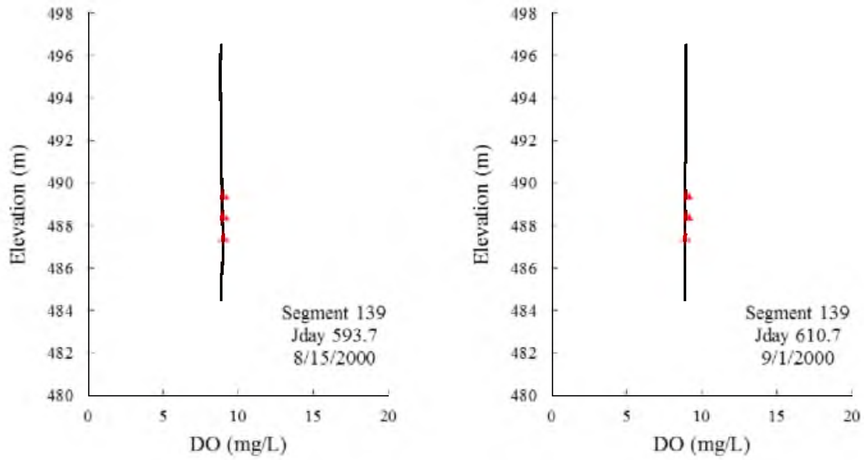


Figure B.23 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 3.3 Miles Upstream of Nine Mile Dam (Segment 139)

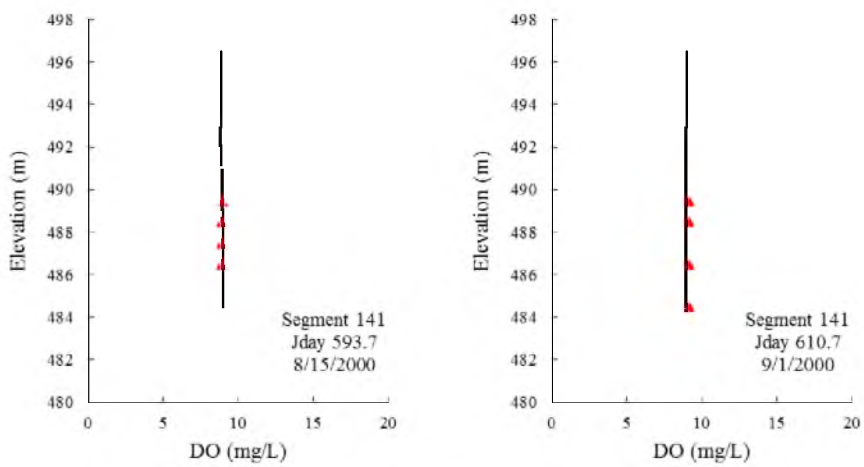


Figure B.24 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 2.8 Miles Upstream of Nine Mile Dam (Segment 141)

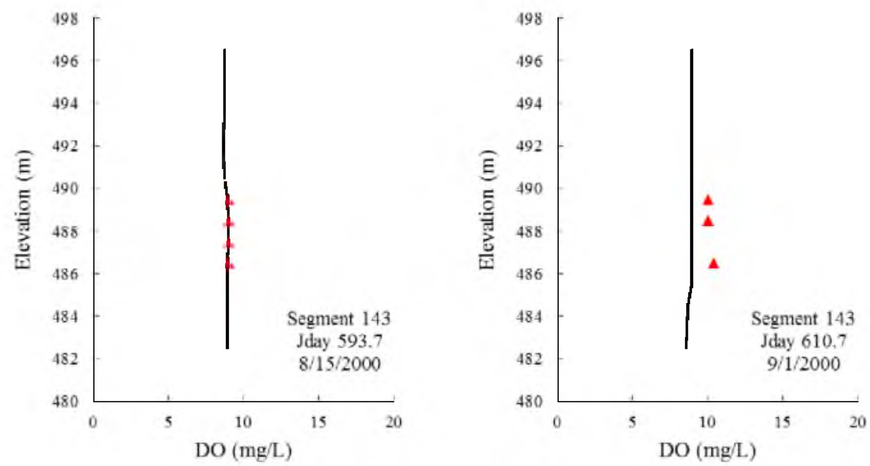


Figure B.25 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 2.1 Miles Upstream of Nine Mile Dam (Segment 143)

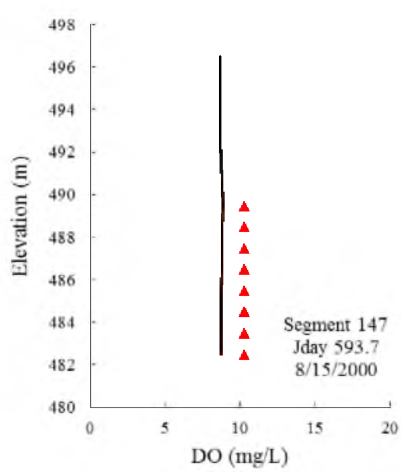


Figure B.26 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 0.8 Miles Upstream of Nine Mile Dam (Segment 147)

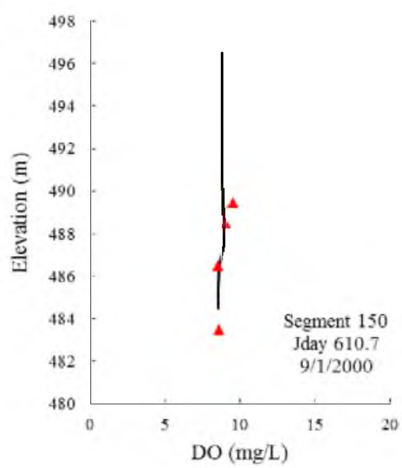


Figure B.27 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 Data (Red Dots) for the Spokane River 0.2 Miles Upstream of Nine Mile Dam (Segment 150)

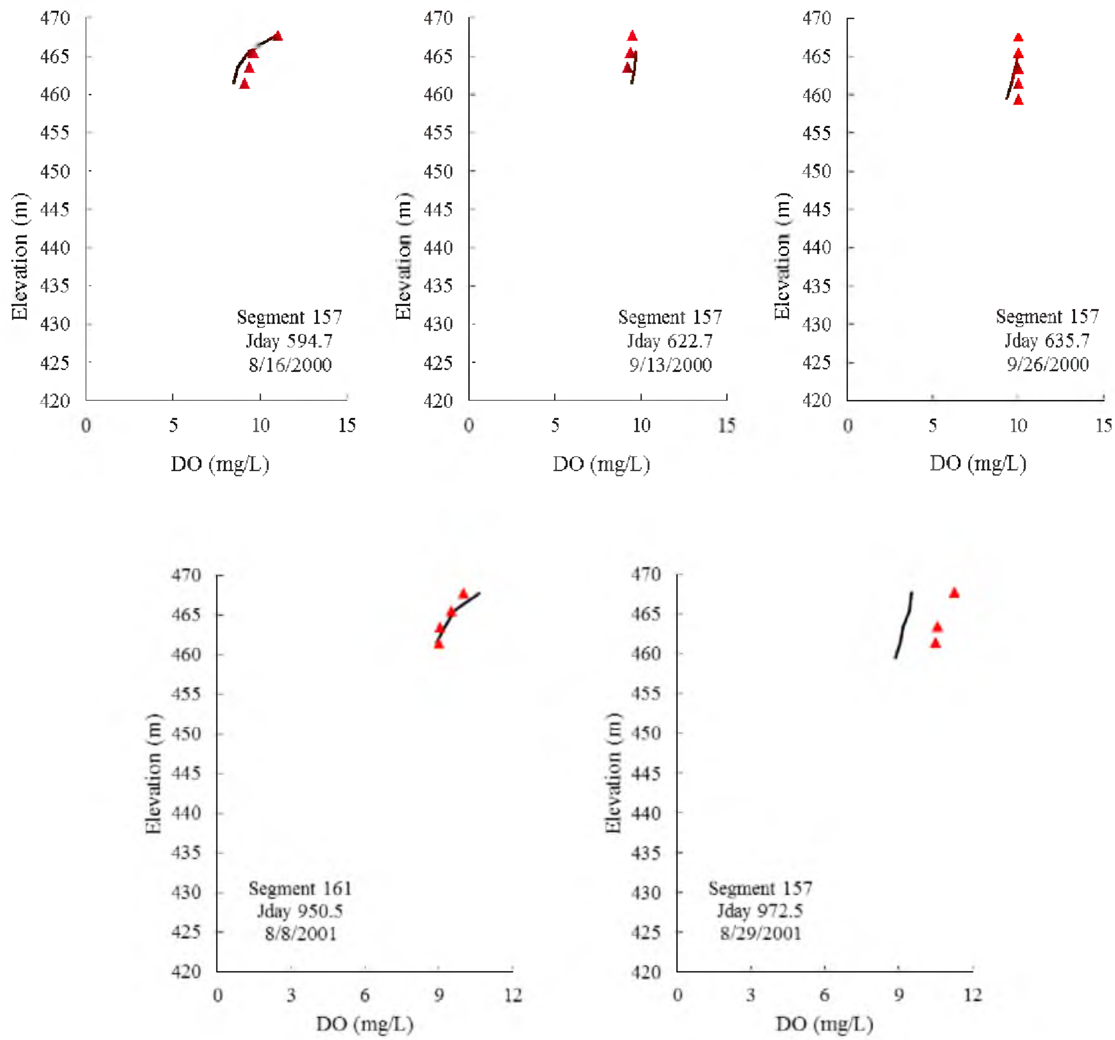


Figure B.28 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 5 (Segment 157)

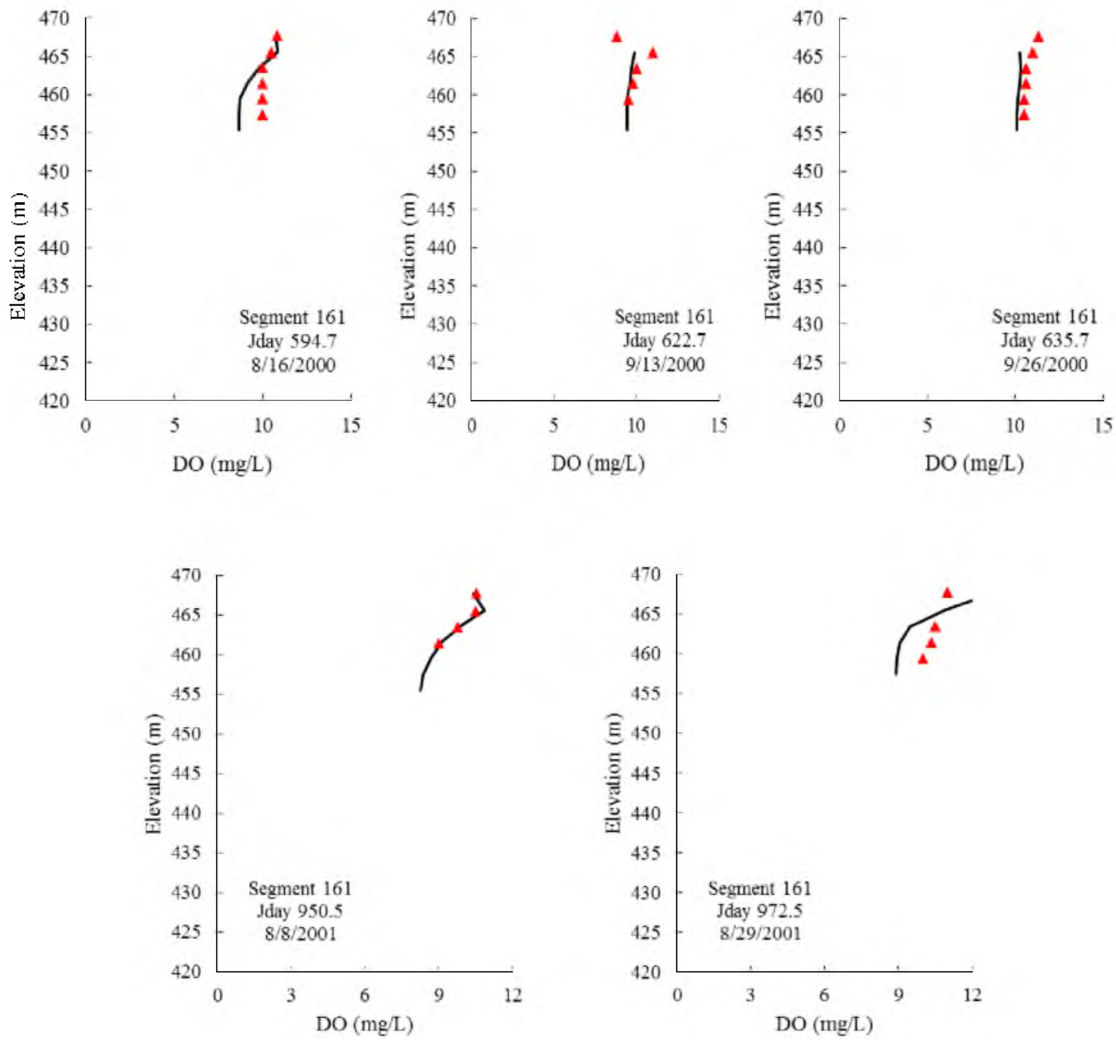


Figure B.29 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 4 (Segment 161)

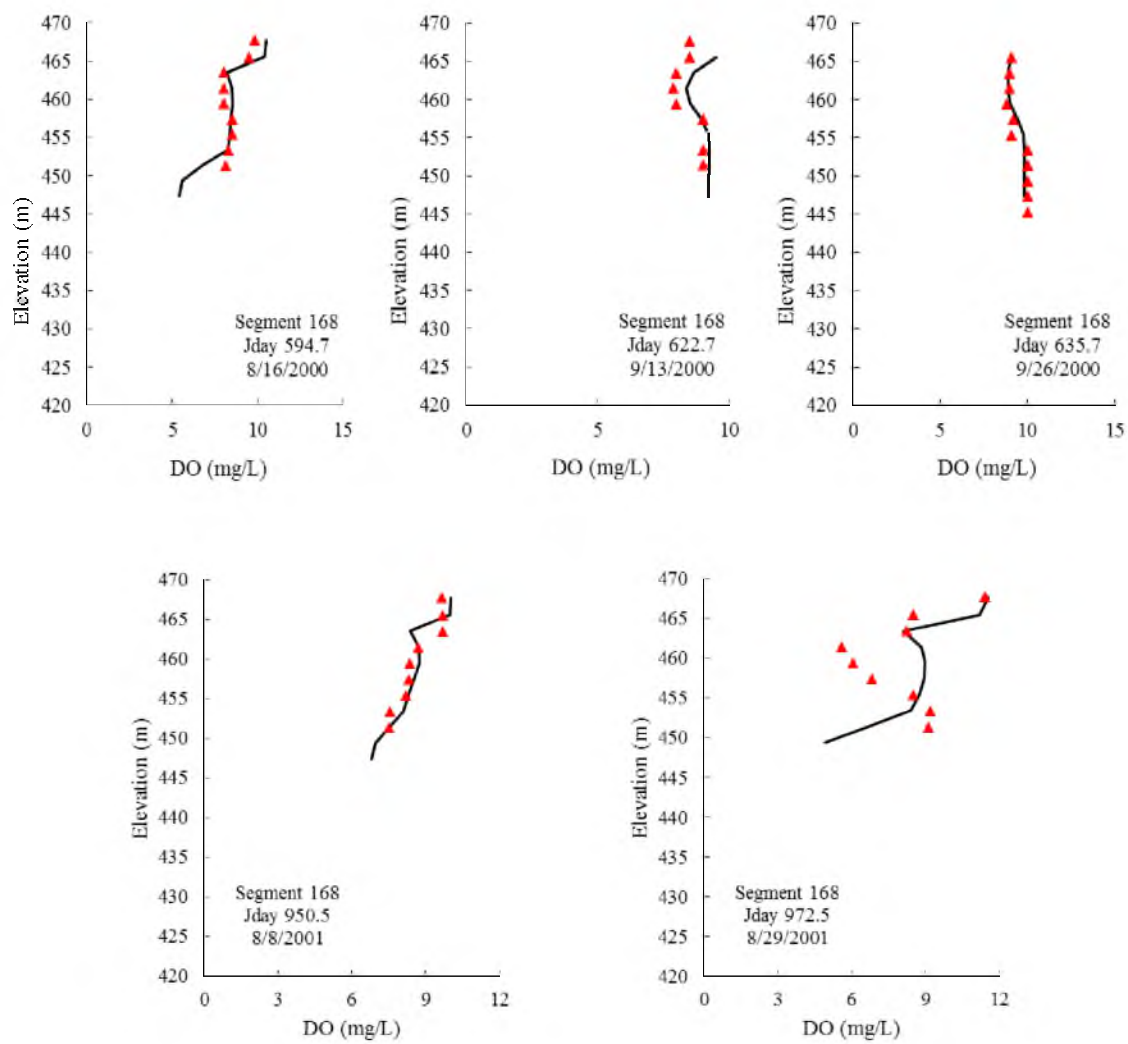


Figure B.30 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 3 (Segment 168)

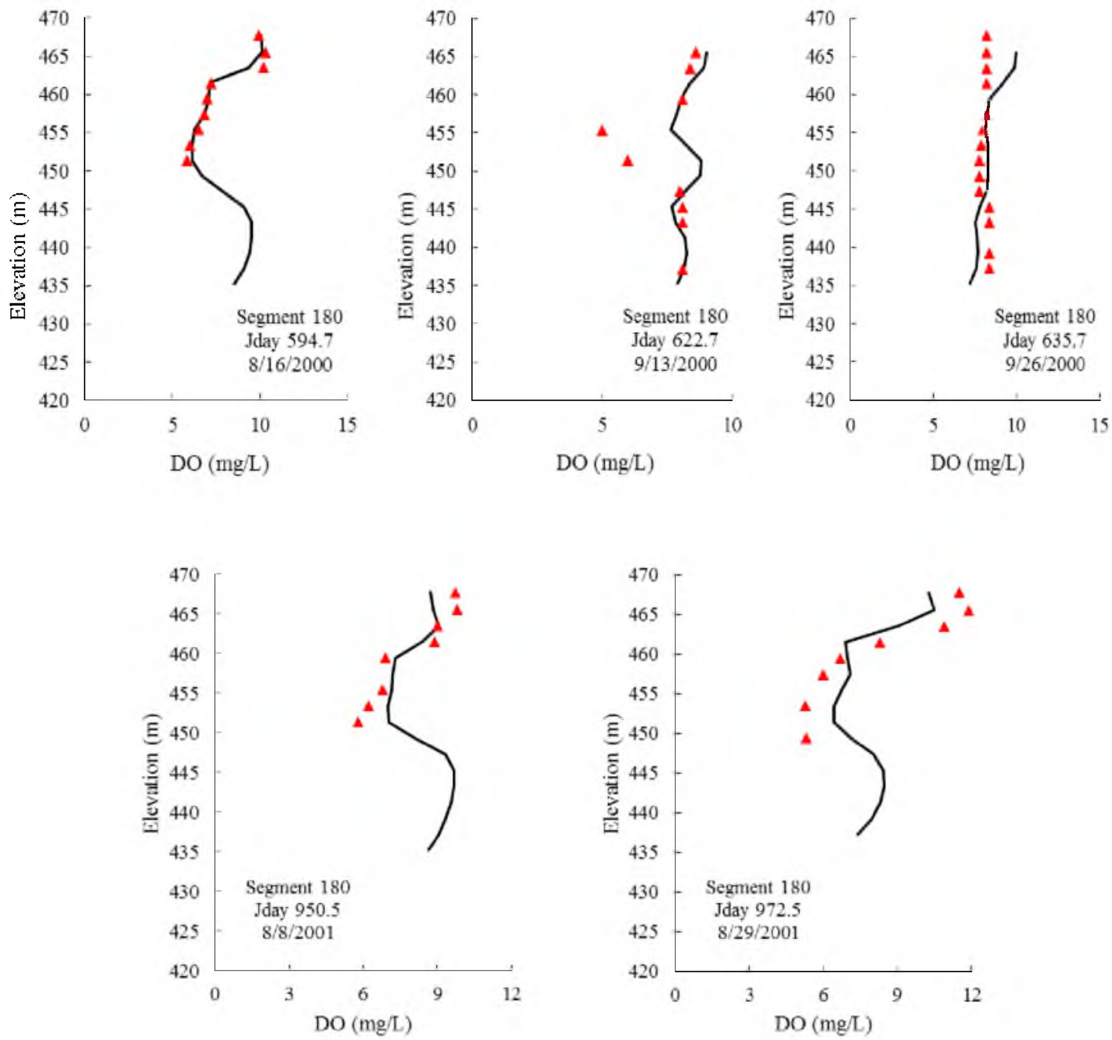


Figure B.31 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 1 (Segment 180)

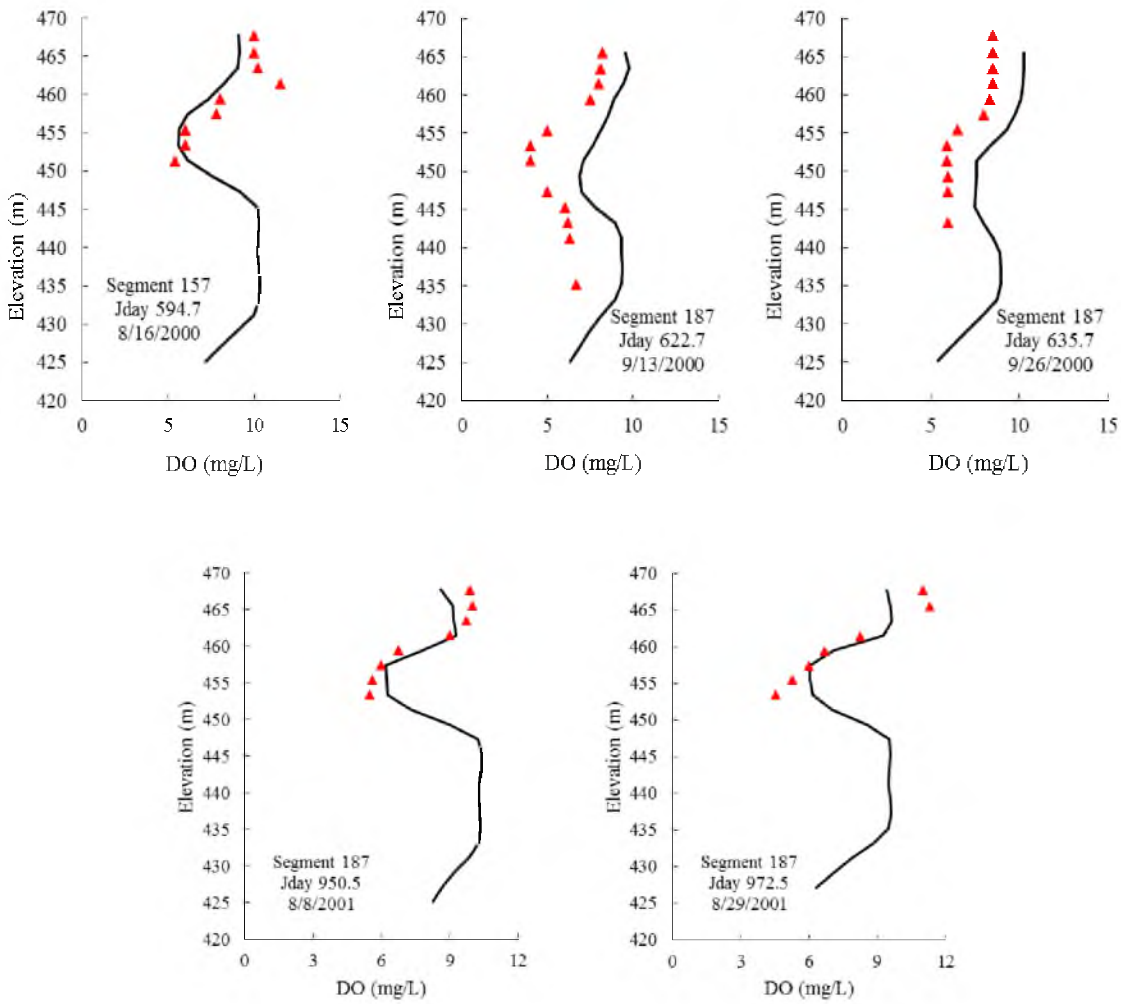


Figure B.32 Comparison of Model Predicted Vertical DO Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 0 (Segment 187)

Table B.6 DO Profile Error Statistics

Location	N, # of data profile comparisons	Error statistics (mg/L)	
		AME	RMS
SPK82.5, Segment 57	1	1.18	1.20
SPK81.6, Segment 60	2	0.63	0.67
SPK81.0, Segment 62	2	0.23	0.26
SPK79.8, Segment 64	2	0.30	0.31
SPK62.0, Segment 135	1	0.02	0.03
SPK61.4, Segment 139	2	0.06	0.07
SPK60.9, Segment 141	2	0.13	0.15
SPK60.2, Segment 143	2	0.54	0.79
SPK58.9, Segment 147	1	1.51	1.52
SPK58.3, Segment 150	1	0.29	0.38
LL5, Segment 157	5	1.08	1.73
LL4, Segment 161	5	0.85	1.28
LL3, Segment 168	5	0.90	1.23
LL2, Segment 174	5	1.02	1.36
LL1, Segment 180	5	1.18	1.76
LL0, Segment 187	5	1.02	1.62

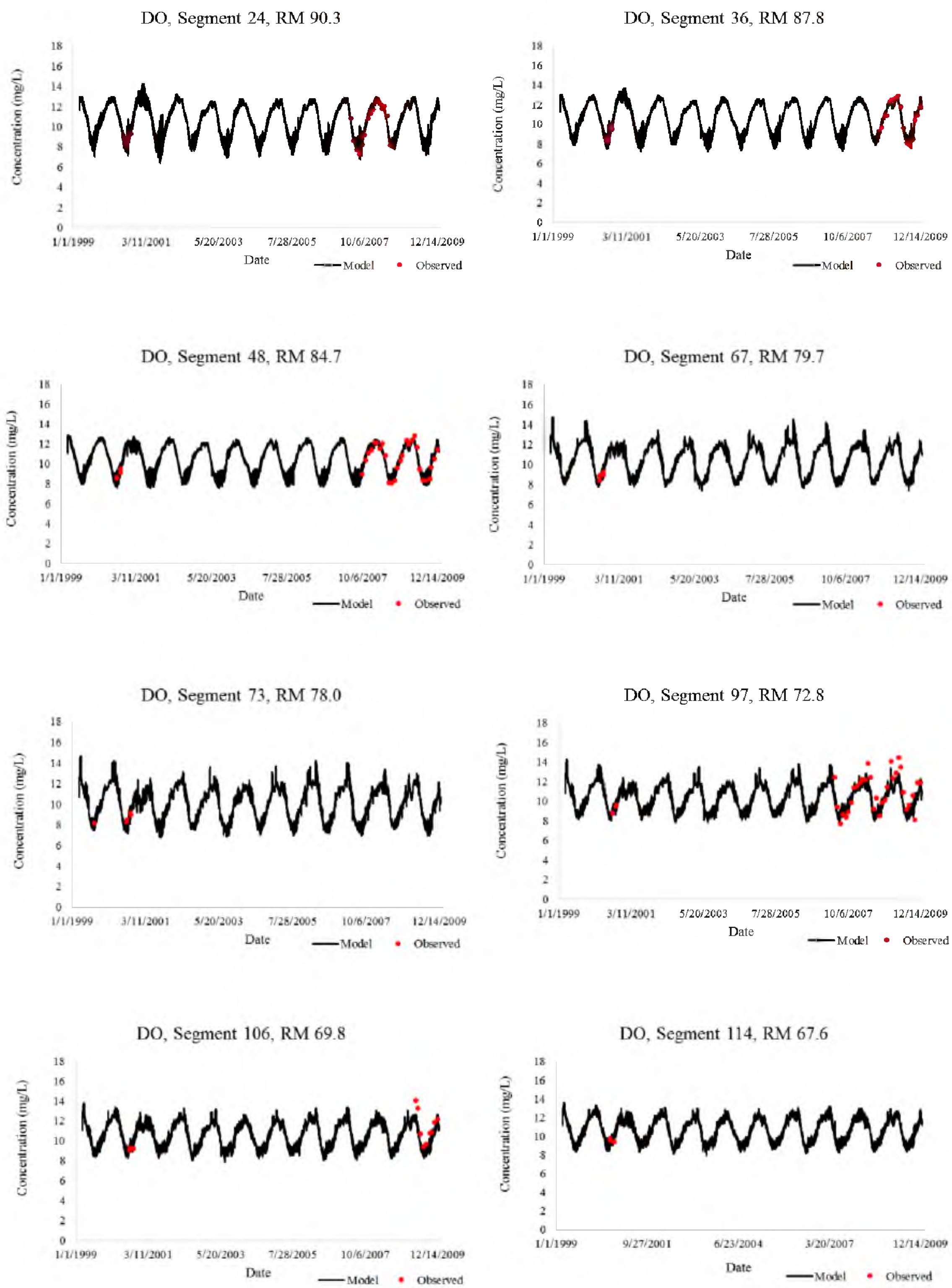


Figure B.33 DO Time Series Comparisons

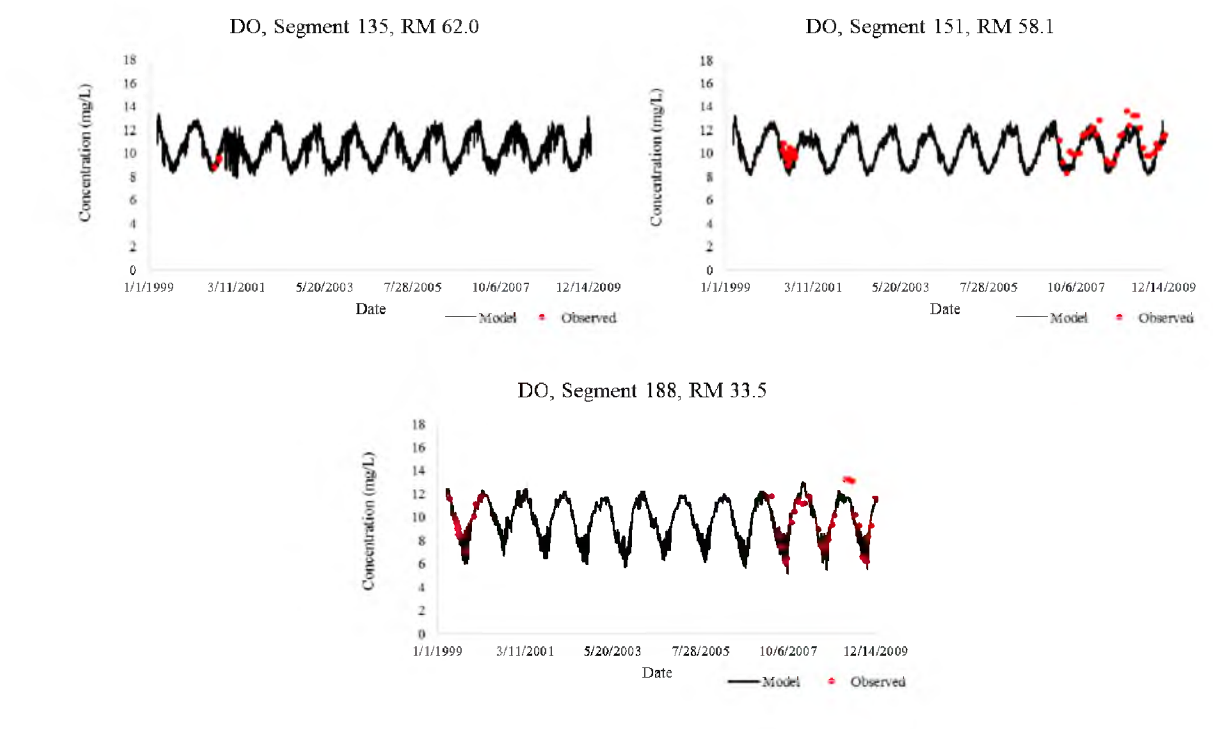


Figure B.33 Continued

Table B.7 DO Time Series Error Statistics

Location	RM	N, # of data	Site error statistics (mg/L)	
			AME	RMS
Baker Road	90.3	19	0.44	0.52
Sullivan Road	87.8	18	0.37	0.48
Plante's Ferry Park	8B.7	27	0.65	1.65
Above Upriver Dam	79.7	4	0.21	0.24
Green Street Bridge	78.0	5	0.24	0.32
Sandifer Bridge	72.6	32	1.05	1.28
Fort Wright Bridge	69.8	13	0.92	1.16
Above Spokane WWTP	67.6	4	0.51	0.52
Riverside State Park	66.0	136	0.67	1.14
Seven Mile Bridge	62.0	4	0.28	0.36
Nine Mile Dam	58.1	58	0.78	0.92
Long Lake Dam	33.5	52	0.83	1.12

Table B.8 DO Time Series Statistics (mg/L)

Location	RM	# of Data	Model Data		Observed Data	
			Mean	Std. Dev.	Mean	Std. Dev.
Baker Road	90.3	19	10.0	1.5	10.0	1.7
Sullivan Road	87.8	18	10.3	1.6	10.4	1.6
Plante's Ferry Park	8B.7	27	10.2	1.5	10.5	1.5
Above Upriver Dam	79.7	4	8.7	0.4	8.9	0.3
Green Street Bridge	78.0	5	8.4	0.5	8.6	0.4
Sandifer Bridge	72.6	32	10.3	1.2	10.9	1.8
Fort Wright Bridge	69.8	13	10.0	1.1	10.8	1.6
Above Spokane WWTP	67.6	4	9.5	0.4	9.5	0.2
Riverside State Park	66.0	136	10.7	1.3	11.0	1.5
Seven Mile Bridge	62.0	4	9.2	0.6	9.2	0.3
Nine Mile Dam	58.1	58	9.8	1.2	10.5	1.2
Long Lake Dam	33.5	52	9.9	1.6	9.4	1.7

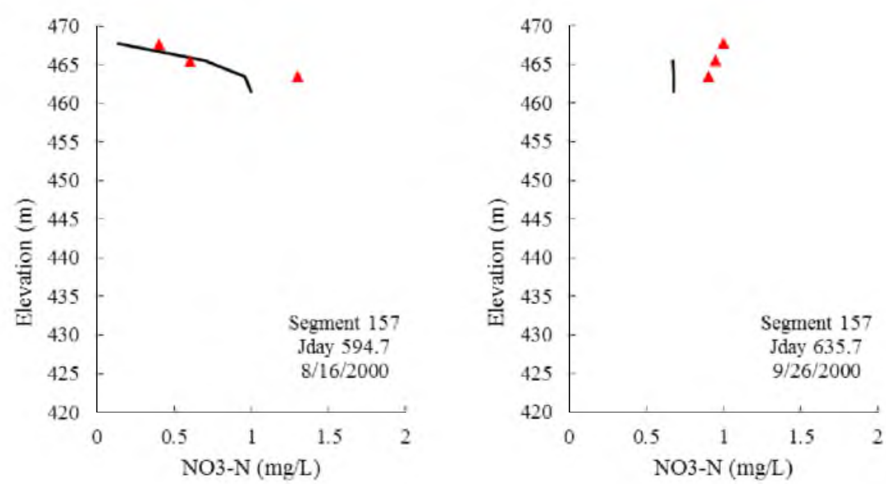


Figure B.34 Comparison of Model Predicted Vertical Nitrite-Nitrate Nitrogen Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 5 (Segment 157)

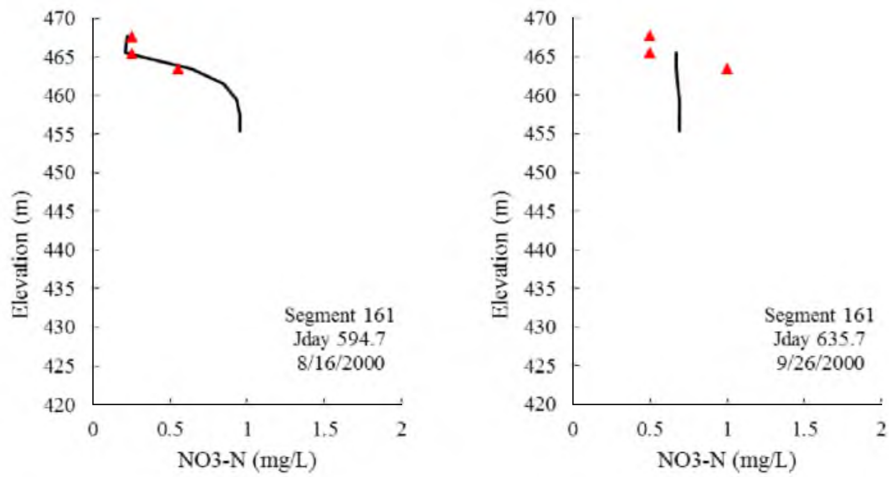


Figure B.35 Comparison of Model Predicted Vertical Nitrite-Nitrate Nitrogen Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 4 (Segment 161)

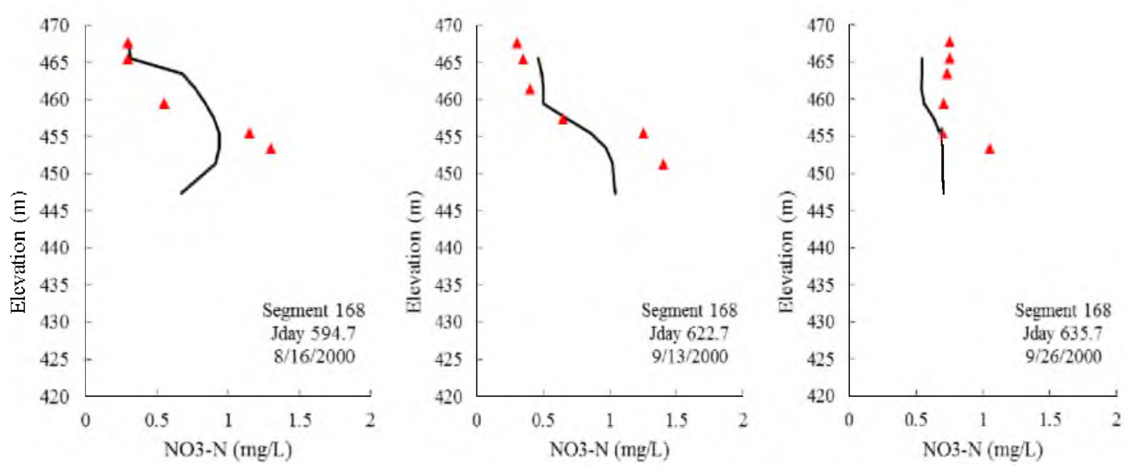


Figure B.36 Comparison of Model Predicted Vertical Nitrite-Nitrate Nitrogen Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 3 (Segment 168)

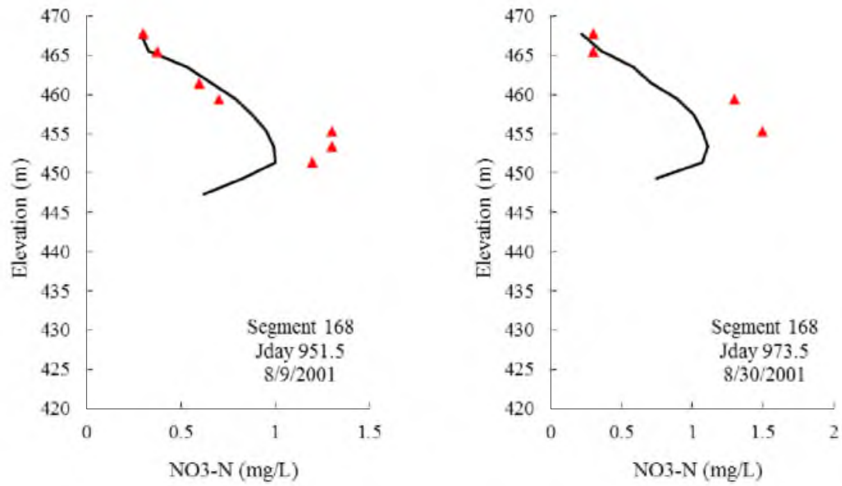


Figure B.36 Continued

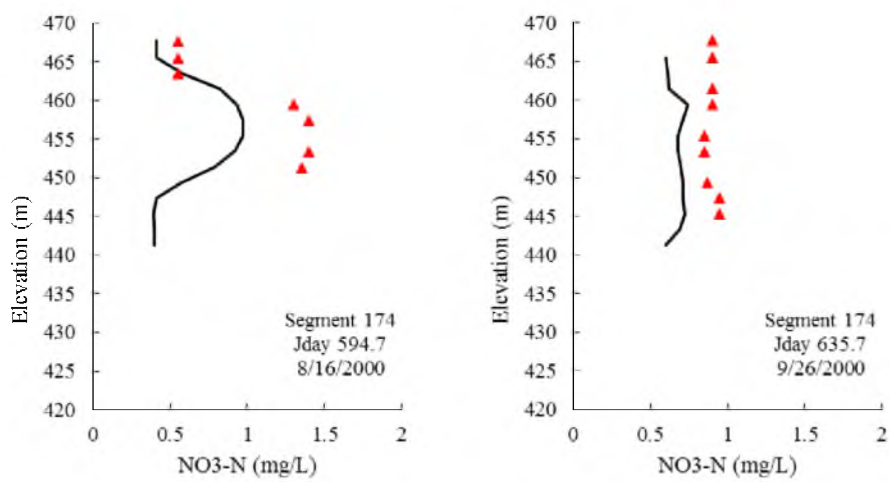


Figure B.37 Comparison of Model Predicted Vertical Nitrite-Nitrate Nitrogen Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 2 (Segment 174)

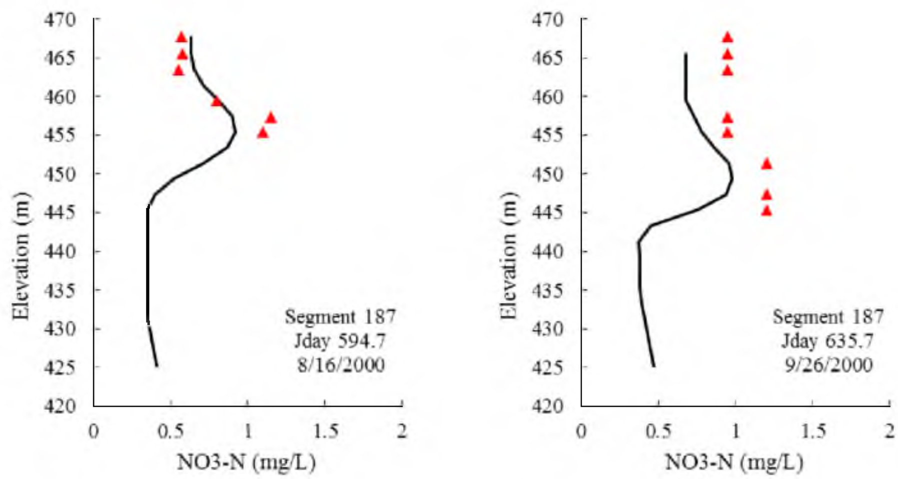


Figure B.38 Comparison of Model Predicted Vertical Nitrite-Nitrate Nitrogen Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 0 (Segment 187)

Table B.9 Nitrate Profile Error Statistics

Location	**N, # of data profile Comparisons**	**Error statistics (mg/L)**	
		AME	**RMS**
LL5, Segment 157	2	0.24	0.25
LL4, Segment 161	2	0.13	0.17
LL3, Segment 168	5	0.18	0.23
LL2, Segment 174	2	0.25	0.29
LL1, Segment 180	5	0.33	0.26
LL0, Segment 187	2	0.19	0.22

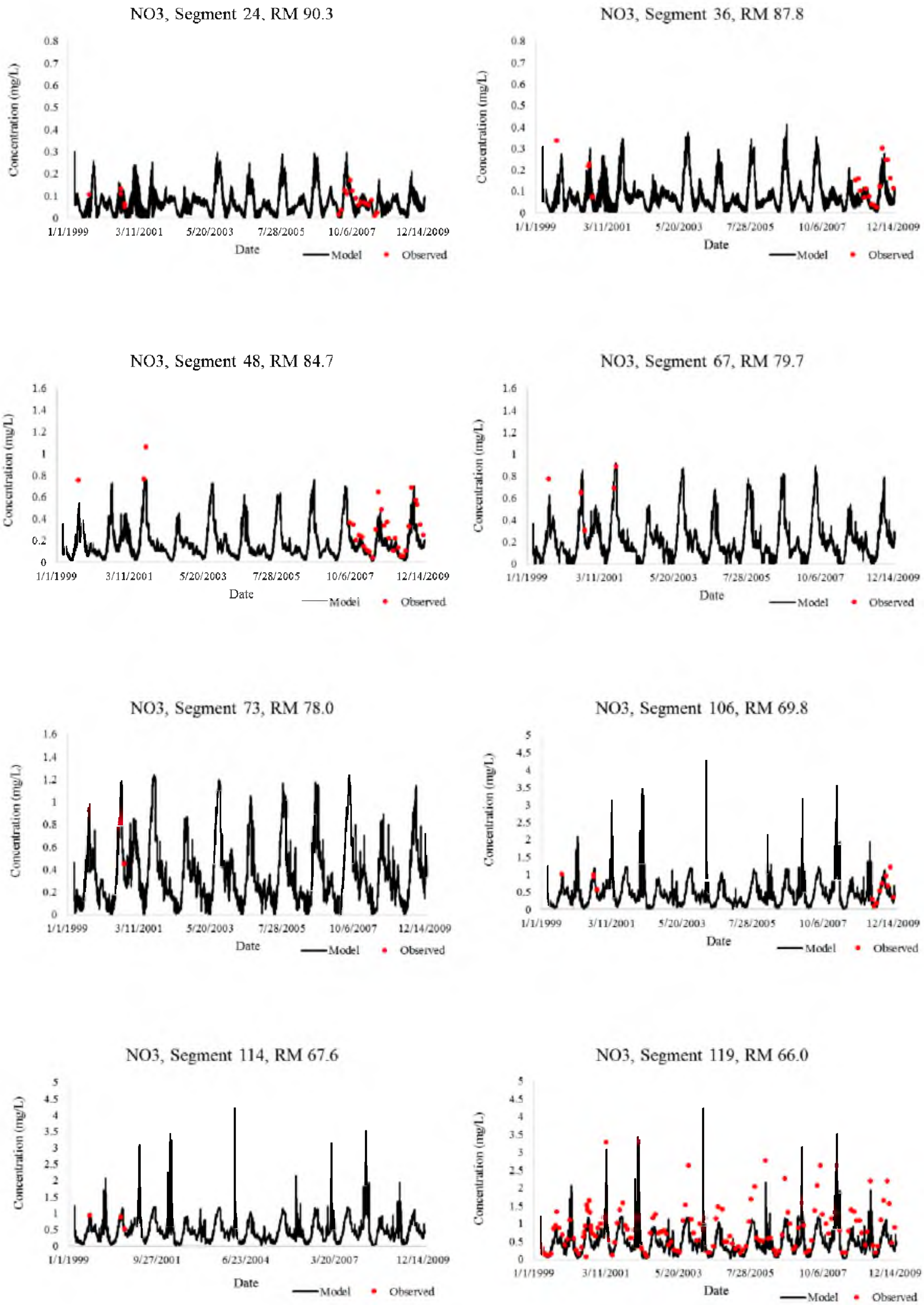


Figure B.39 Nitrate-Nitrogen Time Series Comparisons

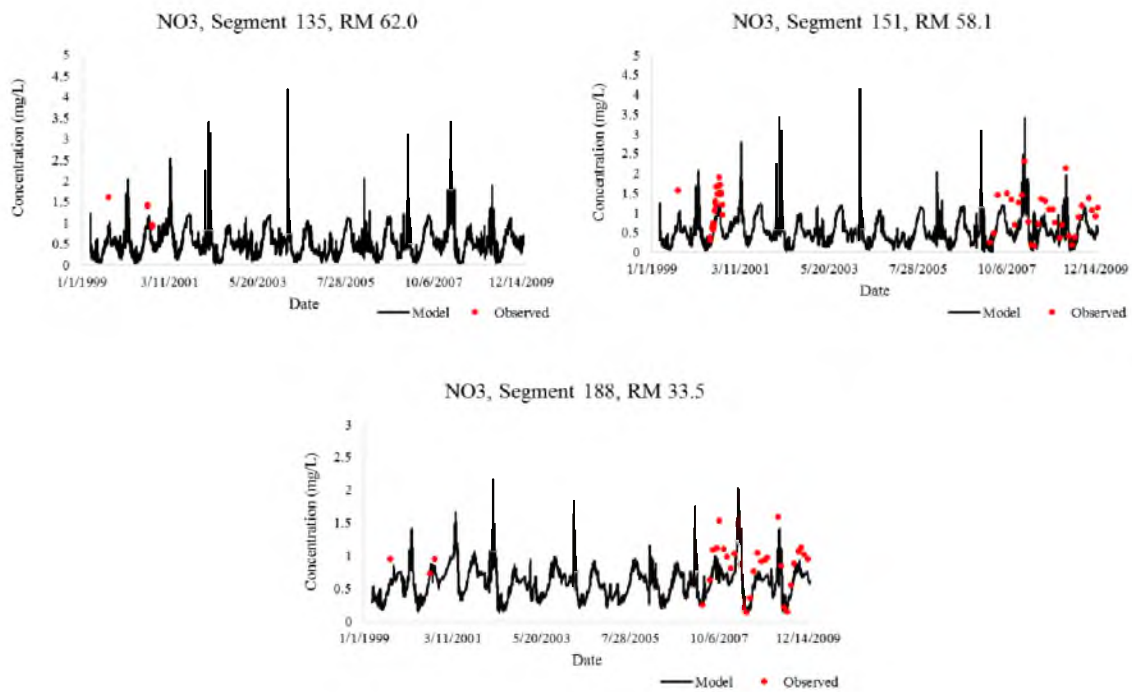


Figure B.39 Continued

Table B.10 Nitrate Time Series Error Statistics

Location	RM	N, # of data	Site error statistics (mg/L)	
			AME	**RMS**
Baker Road	90.3	20	0.038	0.049
Sullivan Road	87.8	20	0.065	0.090
Plante's Ferry Park	8B.7	30	0.131	0.167
Above Upriver Dam	79.7	7	0.123	0.161
Green Street Bridge	78.0	5	0.047	0.063
Sandifer Bridge	72.6	33	0.056	0.082
Fort Wright Bridge	69.8	14	0.185	0.266
Above Spokane WWTP	67.6	5	0.071	0.086
Riverside State Park	66.0	138	0.416	0.564
Seven Mile Bridge	62.0	5	0.533	0.554
Nine Mile Dam	58.1	57	0.466	0.540
Long Lake Dam	33.5	32	0.276	0.325

Table B.11 Nitrate Time Series Statistics (mg/L)

Location	RM	# of Data	Model Data		Observed Data	
			Mean	Std. Dev.	Mean	Std. Dev.
Baker Road	90.3	20	0.063	0.051	0.079	0.041
Sullivan Road	87.8	20	0.082	0.040	0.146	0.088
Plante's Ferry Park	84.7	30	0.211	0.157	0.343	0.250
Above Upriver Dam	79.7	7	0.487	0.212	0.609	0.207
Green Street Bridge	78.0	5	0.654	0.175	0.701	0.206
Sandifer Bridge	72.6	33	0.383	0.259	0.423	0.265
Fort Wright Bridge	69.8	14	0.519	0.224	0.657	0.328
Above Spokane WWTP	67.6	5	0.679	0.150	0.749	0.195
Riverside State Park	66.0	138	0.511	0.419	0.896	0.654
Seven Mile Bridge	62.0	5	0.724	0.162	1.257	0.279
Nine Mile Dam	58.1	57	0.652	0.324	1.034	0.511
Long Lake Dam	33.5	32	0.657	0.230	0.857	0.375

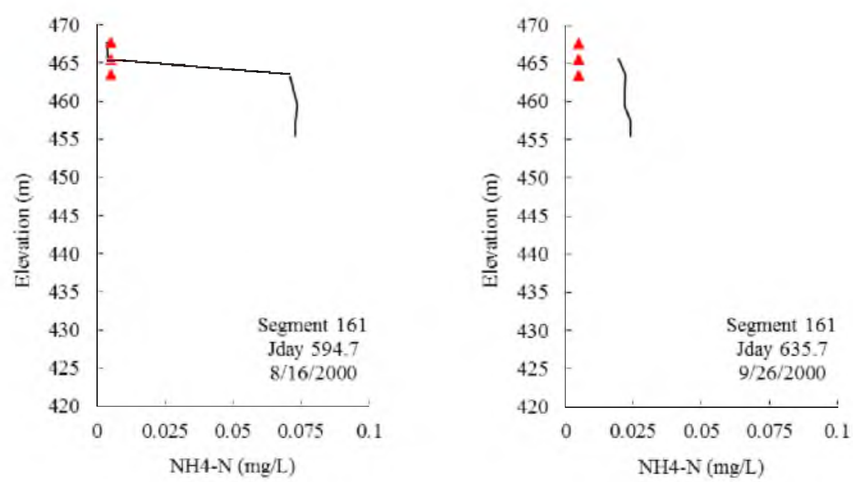


Figure B.40 Comparison of Model Predicted Vertical Ammonia Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 4 (Segment 161)

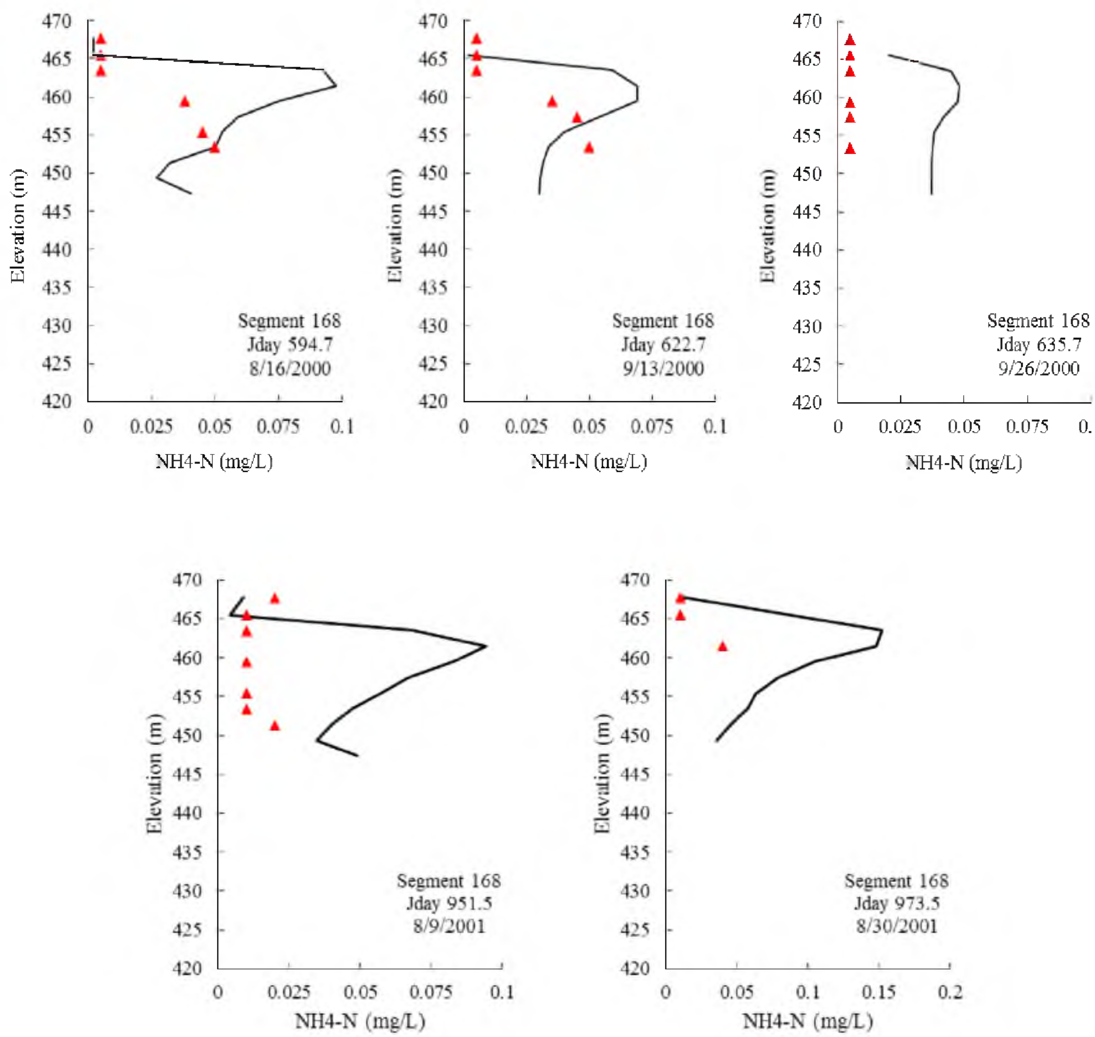


Figure B.41 Comparison of Model Predicted Vertical Ammonia Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 3 (Segment 168)

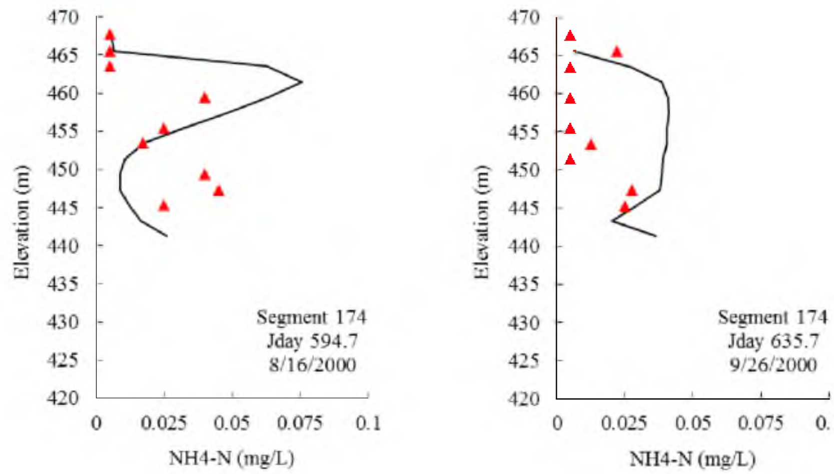


Figure B.42 Comparison of Model Predicted Vertical Ammonia Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 2 (Segment 174)

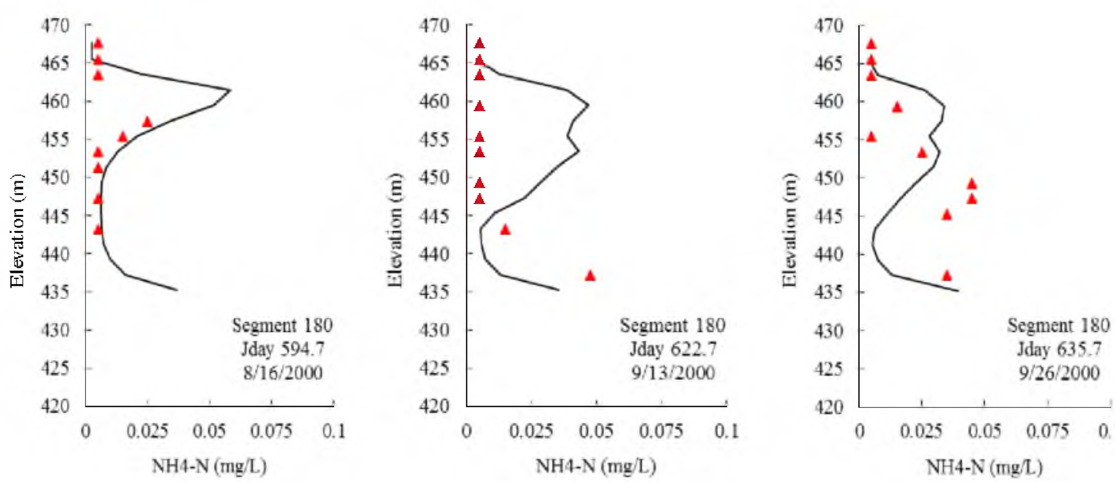
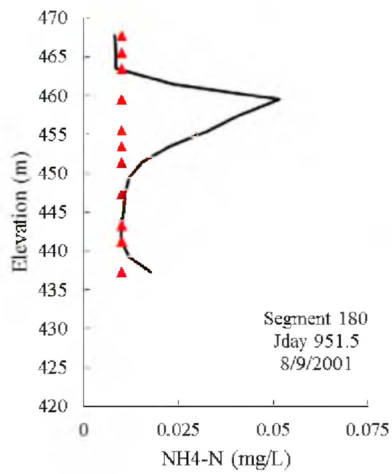


Figure B.43 Comparison of Model Predicted Vertical Ammonia Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 1 (Segment 180)



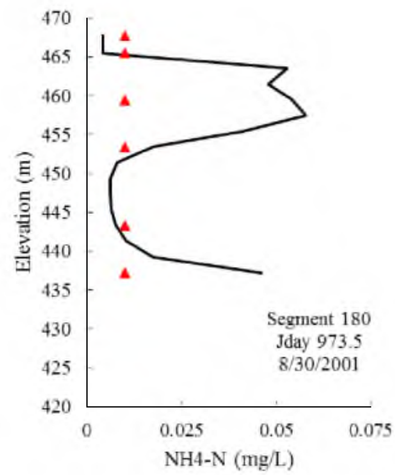


Figure B.43 Continued

Table B.12 Ammonium Profile Error Statistics

Location	N, # of data profile Comparisons	Error statistics (mg/L)	
		AME	RMS
LL4, Segment 161	2	0.020	0.031
LL3, Segment 168	5	0.033	0.043
LL2, Segment 174	2	0.021	0.026
LL1, Segment 180	5	0.013	0.019
LL0, Segment 187	2	0.008	0.014

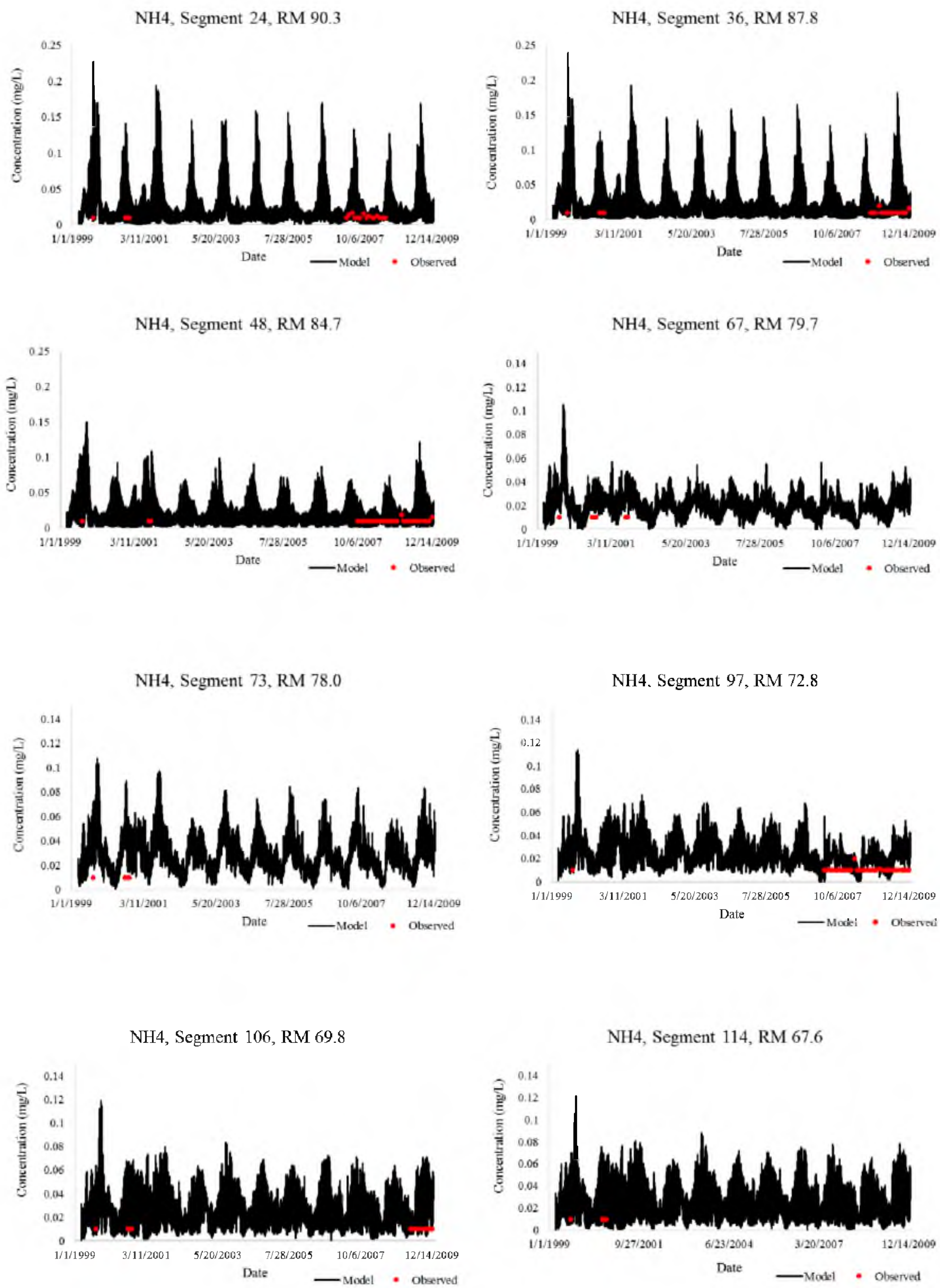


Figure B.44 Ammonia-Nitrogen Time Series Comparisons

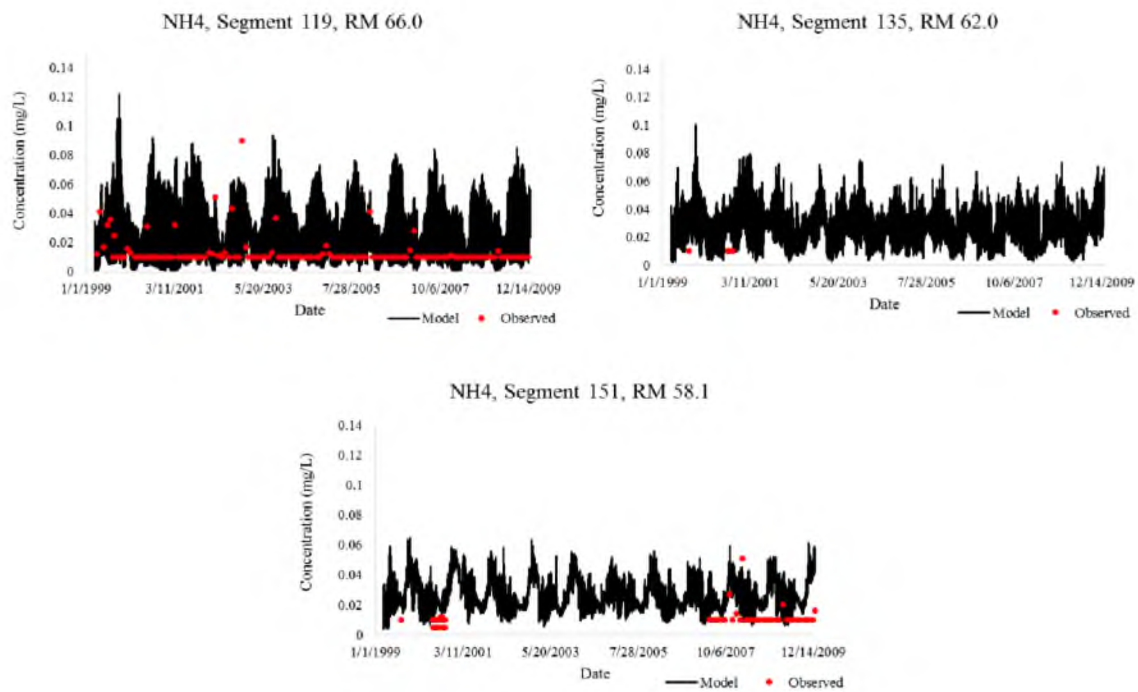


Figure B.44 Continued

Table B.13 Ammonia-Nitrogen Profile Error Statistics

Location	RM	N, # of data	Site error statistics (mg/L)	
			AME	RMS
Baker Road	90.3	20	0.007	0.010
Sullivan Road	87.8	20	0.023	0.034
Plante's Ferry Park	8B.7	30	0.022	0.029
Above Upriver Dam	79.7	7	0.016	0.019
Green Street Bridge	78.0	5	0.019	0.019
Sandifer Bridge	72.6	33	0.009	0.012
Fort Wright Bridge	69.8	14	0.019	0.028
Above Spokane WWTP	67.6	5	0.002	0.002
Riverside State Park	66.0	138	0.014	0.020
Seven Mile Bridge	62.0	5	0.017	0.017
Nine Mile Dam	58.1	60	0.014	0.016
Long Lake Dam	33.5	32	0.010	0.013

Table B.14 Ammonia-Nitrogen Time Series Statistics (mg/L)

Location	RM	# of Data	Model Data		Observed Data	
			Mean	Std. Dev.	Mean	Std. Dev.
Baker Road	90.3	20	0.014	0.009	0.011	0.002
Sullivan Road	87.8	20	0.034	0.025	0.011	0.002
Plante's Ferry Park	84.7	30	0.030	0.022	0.010	0.002
Above Upriver Dam	79.7	7	0.026	0.011	0.010	0.000
Green Street Bridge	78.0	5	0.029	0.005	0.010	0.000
Sandifer Bridge	72.6	33	0.018	0.009	0.010	0.002
Fort Wright Bridge	69.8	14	0.028	0.021	0.010	0.000
Above Spokane WWTP	67.6	5	0.008	0.001	0.010	0.000
Riverside State Park	66.0	138	0.021	0.017	0.013	0.010
Seven Mile Bridge	62.0	5	0.027	0.001	0.010	0.000
Nine Mile Dam	58.1	60	0.024	0.010	0.011	0.006
Long Lake Dam	33.5	32	0.014	0.011	0.015	0.007

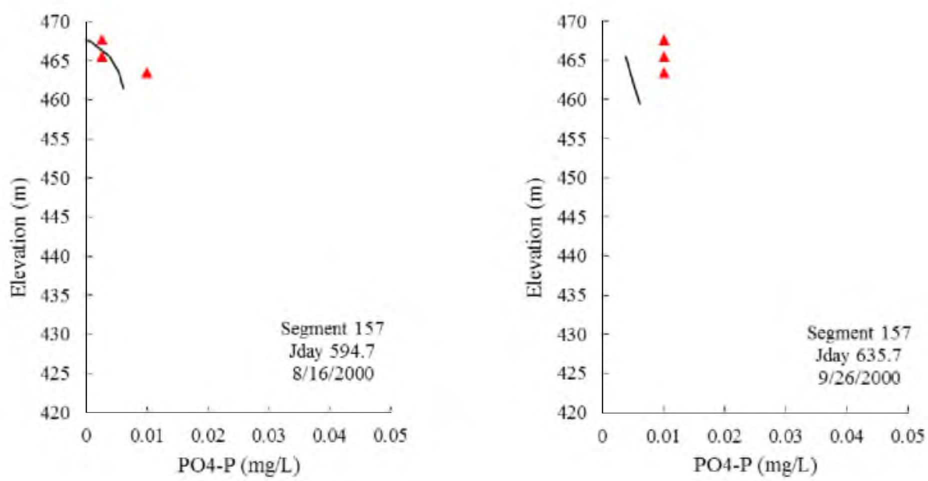


Figure B.45 Comparison of Model Predicted Vertical Soluble Reactive Phosphorus Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 5 (Segment 157)

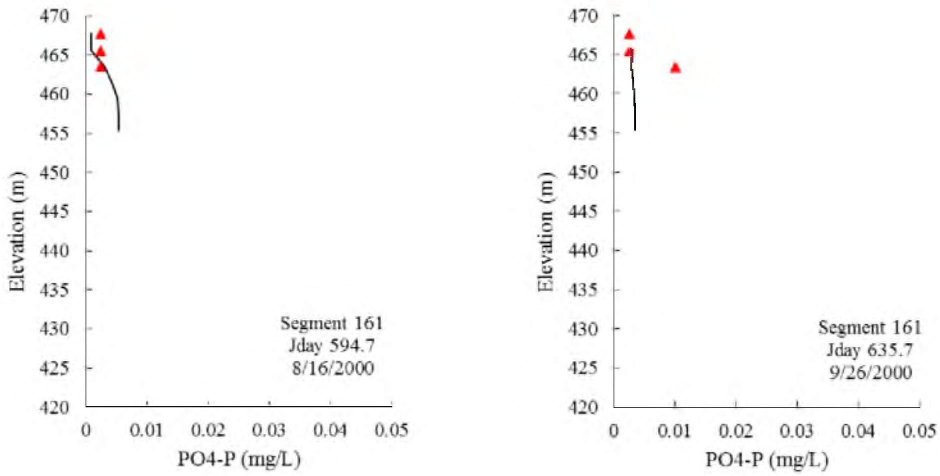


Figure B.46 Comparison of Model Predicted Vertical Soluble Reactive Phosphorus Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 4 (Segment 161)

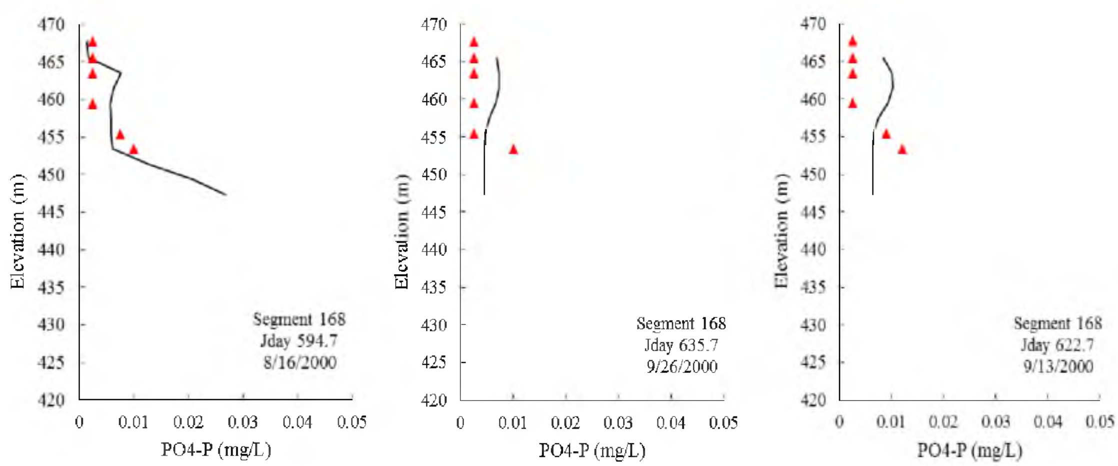


Figure B.47 Comparison of Model Predicted Vertical Soluble Reactive Phosphorus Profiles (Black Line) and 2000 and 2001 Data (Red Dots) for Long Lake at Station 3 (Segment 168)

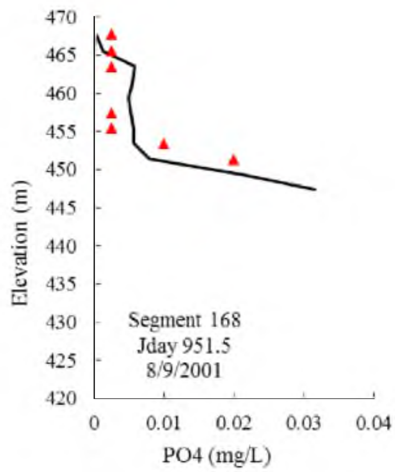


Figure B.47 Continued

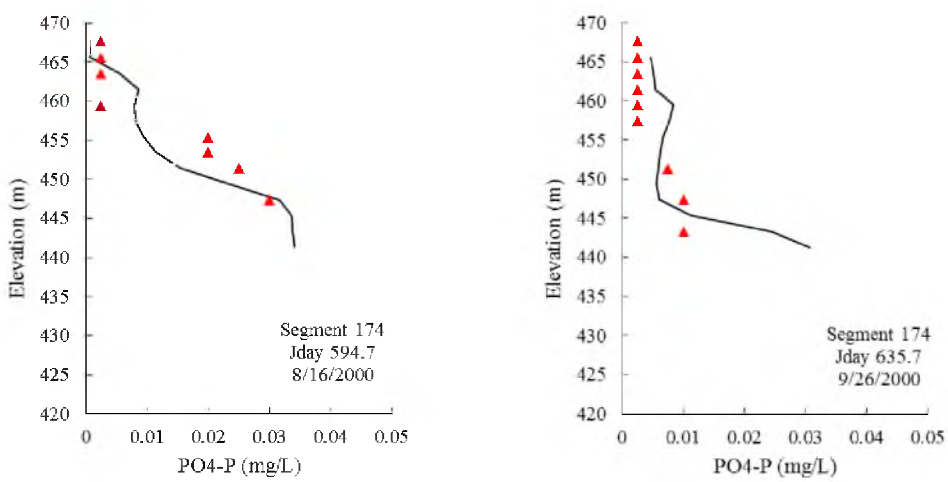


Figure B.48 Comparison of Model Predicted Vertical Soluble Reactive Phosphorus Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 2 (Segment 174)

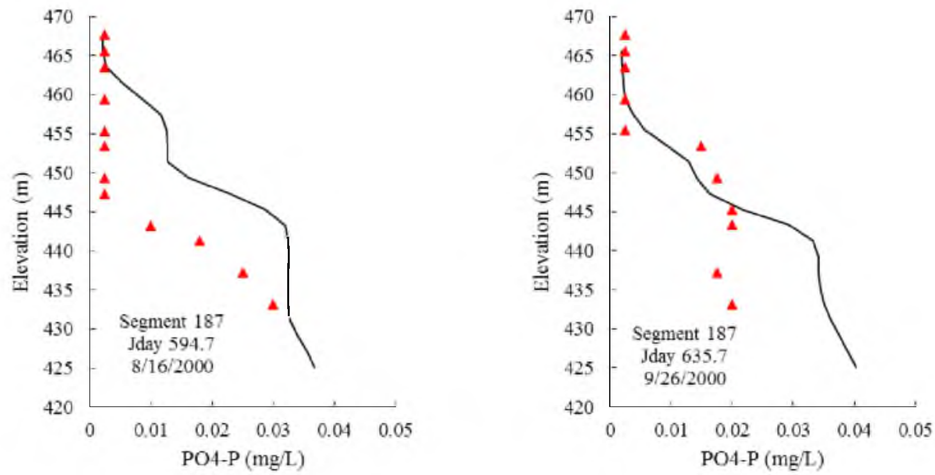


Figure B.49 Comparison of Model Predicted Vertical Soluble Reactive Phosphorus Profiles (Black Line) and 2000 Data (Red Dots) for Long Lake at Station 0 (Segment 187)

Table B.15 Phosphate Profile Error Statistics

Location	N, # of data profile Comparisons	Error statistics (mg/L)	
		AME	RMS
LL5, Segment 157	2	0.004	0.004
LL4, Segment 161	2	0.002	0.003
LL3, Segment 168	5	0.004	0.004
LL2, Segment 174	2	0.005	0.006
LL1, Segment 180	5	0.005	0.007
LL0, Segment 187	2	0.007	0.010

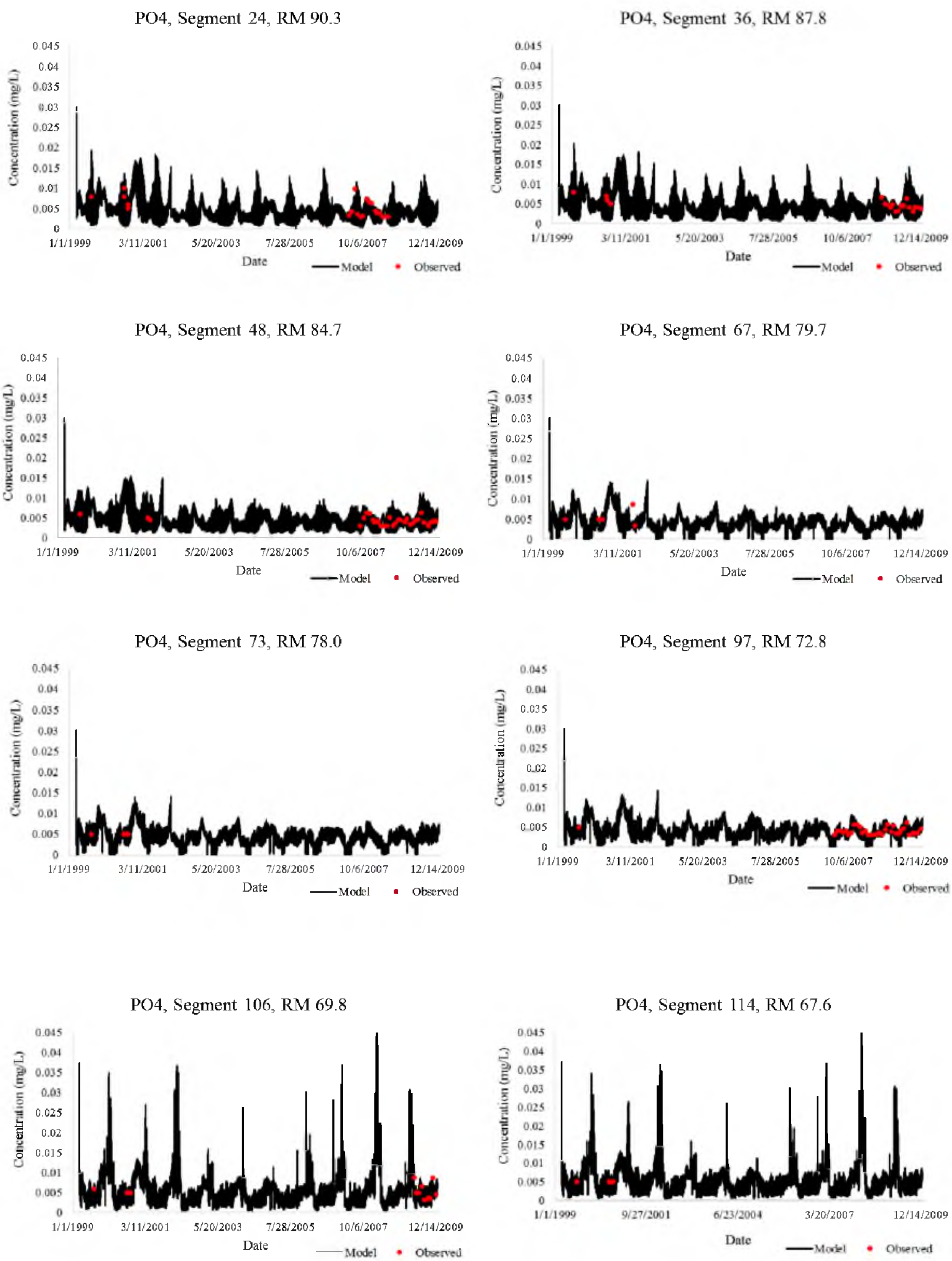


Figure B.50 Phosphate Time Series Comparisons

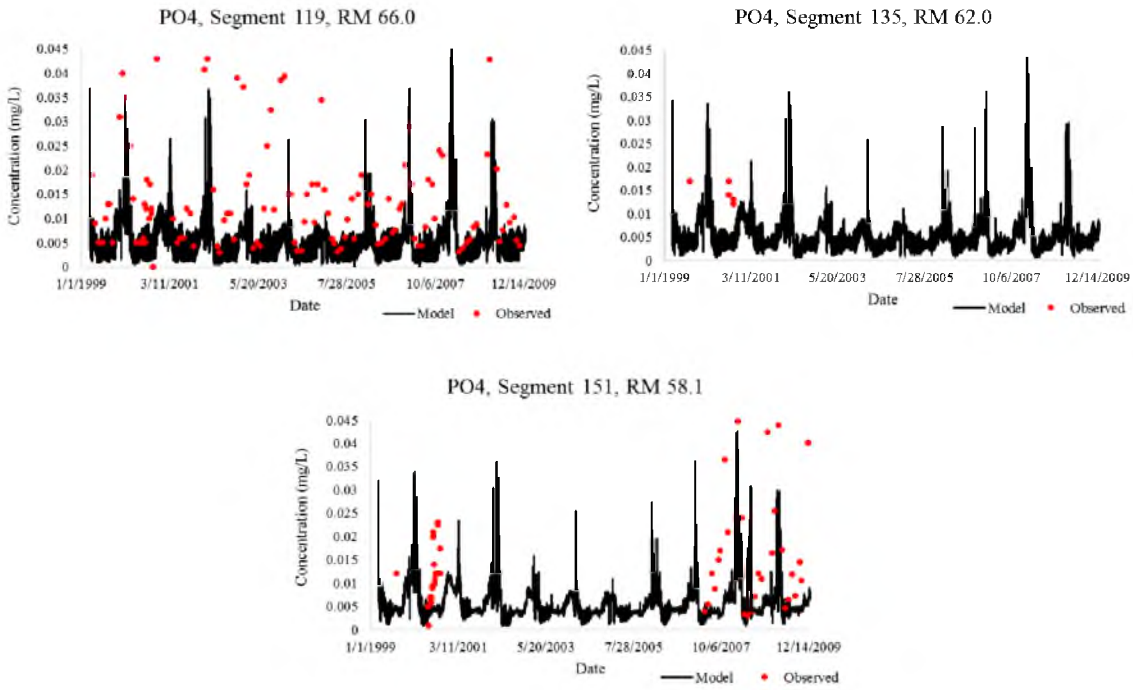


Figure B.50 Continued

Table B.16 Phosphate Time Series Error Statistics

Location	RM	N, # of data	Site error statistics (mg/L)	
			AME	RMS
Baker Road	90.3	20	0.003	0.004
Sullivan Road	87.8	20	0.003	0.003
Plante’s Ferry Park	84.7	30	0.002	0.002
Above Upriver Dam	79.7	7	0.003	0.003
Green Street Bridge	78.0	5	0.002	0.002
Sandifer Bridge	72.6	33	0.001	0.002
Fort Wright Bridge	69.8	14	0.003	0.003
Above Spokane WWTP	67.6	5	0.003	0.003
Riverside State Park	66.0	137	0.015	0.022
Seven Mile Bridge	62.0	5	0.011	0.012
Nine Mile Dam	58.1	55	0.009	0.013
Long Lake Dam	33.5	32	0.008	0.013

Table B.17 Phosphate Time Series Statistics (mg/L)

Location	RM	# of Data	Model Data		Observed Data	
			Mean	Std. Dev.	Mean	Std. Dev.
Baker Road	90.3	20	0.003	0.002	0.005	0.002
Sullivan Road	87.8	20	0.005	0.003	0.005	0.001
Plante's Ferry Park	84.7	30	0.005	0.002	0.004	0.001
Above Upriver Dam	79.7	7	0.003	0.001	0.005	0.001
Green Street Bridge	78.0	5	0.003	0.001	0.005	0.000
Sandifer Bridge	72.6	33	0.004	0.002	0.004	0.001
Fort Wright Bridge	69.8	14	0.004	0.003	0.005	0.002
Above Spokane WWTP	67.6	5	0.002	0.000	0.005	0.000
Riverside State Park	66.0	137	0.005	0.005	0.020	0.019
Seven Mile Bridge	62.0	5	0.003	0.000	0.015	0.002
Nine Mile Dam	58.1	55	0.006	0.004	0.014	0.010
Long Lake	33.5	32	0.008	0.003	0.015	0.013

Table B.18 Typical Model-Data Calibration Errors from Literature

Water quality variable	Calibration AME	Reference	Waterbody
Ammonia	0.02-0.04 mg/L	Berger et al. (2003)	Long Lake Spokane River (WA)
Ammonia	0.02 mg/L	Sullivan and Rounds (2005)	Hagg Lake (OR)
Ammonia-N	0.007-0.039 mg/L	Berger et al. (2001)	Willamette River (OR)
Ammonia-N	0.004-0.048 mg/L	Berger et al. (2002)	Long Lake Spokane River (WA)
Ammonia-N	0.016-0.065 mg/L	Annear et al. (2005)	Spokane River (ID)
Chlorophyll a	0.6 μg/L	Annear et al. (2006)	Pend Oreille (WA)
Chlorophyll a	2.2-25 μg/L	Berger et al. (2001)	Willamette River (OR)
Chlorophyll a	2.4 μg/L	Sullivan and Rounds (2005)	Hagg Lake (OR)
Chlorophyll a	0.87-1.12 μg/L	Annear et al. (2005)	Spokane River (ID)
Dissolved oxygen	0.9 mg/L	Wells et al. (2000)	Cooper Creek (OR) Reservoir Sutherlin
Dissolved oxygen	1.6-2.1 mg/L	Berger et al. (2003)	Long Lake Spokane River (WA)
Dissolved oxygen	0.6-1.1 mg/L	Berger and Wells (2005)	Lake Whatcom (WA)
Dissolved oxygen	0.2 mg/L	Annear et al. (2006)	Pend Oreille (WA)
Dissolved oxygen	0.22-0.61 mg/L	Annear et al. (2005)	Spokane River (ID)
Dissolved oxygen	0.2-2.2 mg/L	Berger et al. (2001)	Willamette River (OR)
Dissolved oxygen	0.26-1.82 mg/L	Berger et al. (2002)	Long Lake Spokane River (WA)
Dissolved oxygen	0.4-0.73 mg/L	Sullivan and Rounds (2005)	Hagg Lake (OR)
Dissolved oxygen	0.53-1.46 mg/L	Hanna and Campbell (2000)	Klamath River (OR-CA)
Dissolved PO_4-P	0.005-0.012 mg/L	Berger et al. (2001)	Willamette River (OR)
PO_4-P	0.002-0.006 mg/L	Berger et al. (2002)	Long Lake Spokane River (WA)
PO_4-P	0.002-0.003 mg/L	Annear et al. (2005)	Spokane River (ID)
Nitrate + Nitrite	0.1-0.52 mg/L	Berger et al. (2003)	Long Lake Spokane River (WA)

Table B.18 Continued

Water quality variable	**Calibration AME**	**Reference**	**Waterbody**
Nitrate + Nitrite	0.068-0.234 mg/L	Berger et al. (2001)	Willamette River (OR)
Nitrate + Nitrite	0.047-0.051 mg/L	Annear et al. (2005)	Spokane River (ID)
Nitrate + Nitrite	0.10-0.42 mg/L	Berger et al. (2002)	Long Lake Spokane River (WA)
Temperature	0.9 °C	Wells et al. (2000)	Cooper Creek (OR) Reservoir Sutherlin
Temperature	0.7-1.01 °C	Berger et al. (2003)	Long Lake Spokane River (WA)
Temperature	0.44-0.67 °C	Berger and Wells (2005)	Lake Whatcom (WA)
Temperature	0.40-0.79 °C	Annear et al. (2005)	Spokane River (ID)
Temperature	0.2-0.6 °C	Annear et al. (2006)	Pend Oreille (WA)
Temperature	0.28-3.49 °C	Berger et al. (2002)	Long Lake Spokane River (WA)
Temperature	0.4-2.1 °C	Berger et al. (2001)	Willamette River (OR)
Temperature	0.68 °C	Sullivan and Rounds (2005)	Hagg Lake (OR)
Temperature	0.5 - 2.29 °C	Hanna and Campbell (2000)	Klamath River (OR-CA)
Total P	0.008-0.025 mg/L	Berger et al. (2001)	Willamette River (OR)
Total P	0.006-0.012 mg/L	Sullivan and Rounds (2005)	Hagg Lake (OR)
Total P	0.006-0.008 mg/L	Annear et al. (2005)	Spokane River (ID)

REFERENCES

Abatzoglou, J.T. and T.J. Brown, 2012. A Comparison of Statistical Downscaling Methods Suited for Wildfire Applications. *International Journal of Climatology* 32(5): 772-780.

Alkalinity Tech Note. Retrieved from: http://www.tocsystemsinc.com/products/alk/docs/alk%20tech%20note.pdf

Andersen, H.E., B. Kronvang, S.E. Larsen, C.C. Hoffmann, T.S. Jensen and E.K. Rasmussen, 2006. Climate-change Impacts on Hydrology and Nutrients in a Danish Lowland River Basin. *Science of the Total Environment* 365(1-3): 223-237.

Andersson, A., H.E.M. Meier, M. Ripszam, O. Rowe, J. Wikner, P. Haglund, K. Eilola, C. Legrand, D. Figueroa, J. Paczkowska, E. Lindehoff, M. Tysklind and R. Elmgren, 2015. Projected Future Climate Change and Baltic Sea Ecosystem Management. *AMBIO: A Journal of the Human Environment* 44(3): 345-356.

Andrews, W.J., M.F. Becker, S.J. Smith and R.L. Tortorelli, 2009. Summary of Surface-water Quality Data from the Illinois River Basin in North Oklahoma, 1970–2007. U.S. Geological Survey Scientific Investigations Report 2009–5182, (revised April 2010.) 39 p. http://pubs.usgs.gov/sir/2009/5182/pdf/sir2009-5182.pdf

Annear, R., C. Berger and S. Wells, 2001. Upper Spokane River Model: Boundary Conditions and Model Setup, 1991 and 2000. Technical Report EWR-4-01, Department of Civil Engineering, Portland State University, Portland, Oregon. URL: http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1149&context=cengin_fac

Annear, R., C. Berger and S. Wells, 2006. Idaho Pend Oreille River Model Development and Calibration. Technical Report EWR-02-06, Department of Civil and Environmental Engineering, Portland State University, Portland, Oregon. https://www.deq.idaho.gov/media/571506-pend_oreille_river_model_entire.pdf

Annear, R., S. Wells and C. Berger, 2005. Upper Spokane River Model in Idaho: Boundary Conditions and Model Setup and Calibration for 2001 and 2004. Technical Report EWR-02-05. Civil and Environmental Engineering, Portland State University. URL: http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1159&context=cengin_fac

Anthony, S.S., C.D. Hunt, A.M.M. Brasher, L.D. Miller and M.S. Tomlinson, 2004. Water

Quality on the Island of Oahu, Hawaii: Reston, Va. U.S. Geological Survey Circular 2004–1239, 37p. http://pubs.usgs.gov/circ/2004/1239/ pdf/circular1239.pdf

Aquarium-fertilizer.com. Retrieved from: http://aquarium-fertilizer.com/nitrate-no3-and-phosphate-po4-dont-cause-algae-ammonia-does

Arheimer, B., J. Andréasson, S. Fogelberg, H. Johnsson, C.B. Pers and K. Persson, 2005. Climate Change Impact on Water Quality: Model Results from Southern Sweden. *AMBIO: A Journal of the Human Environment* 34(7): 559-566.

Arnell, N.W., 2004. Climate Change and Global Water Resources: SRES Emissions and Socio-Economic Scenarios. *Global Environmental Change* 14(1): 31–52.

Arnold, J.G., D.N. Moriasi, P.W. Gassman, K.C. Abbaspour, M.J. White, R. Srinivasan, C. Santhi, R.D. Harmel, A. van Griensven, M.W. Van Liew, N. Kannan and M.K. Jha, 2012. SWAT: Model Use, Calibration, and Validation. *American Society of Agricultural and Biological Engineers* 55(4): 1494-1508.

Arvola, L., G. George, D.M. Livingstone, M. Järvinen, T. Blenckner, M.T. Dokulil, E. Jennings, C.N. Aonghusa, P. Nõges, T. Nõges and G.A. Weyhenmeyer, 2010. The Impact of the Changing Climate on the Thermal Characteristics of Lakes. In. The Impact of Climate Change on European Lakes. Aquatic Ecology Series 4, G. George (ed), Springer Science+Business Media B.V. pp. 85-101.

Backlund, P., D. Schimel, A. Janetos, J. Hatfield, M.G. Ryan, S.R. Archer and D. Lettenmaier, 2008. Introduction. In: The Effects of Climate Change on Agriculture, Land Resources, Water Resources, and Biodiversity in the United States. A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research. Washington, DC., USA, 362 pp. http://www.usda.gov/oce/climate_change/SAP4_3/ CCSPFinalReport.pdf

Barber, M., A. Hossain, J. Covert and G.J. Gregory, 2009. Augmentation of Seasonal Low Stream Flows by Artificial Recharge in the Spokane Valley-Rathdrum Prairie Aquifer of Idaho and Washington, USA. *Hydrogeology Journal* 17(6): 1459-1470.

Barber, M.E., M.A. Hossain, C.J. Poor, C. Shelton, L. Garcia and M. McDonald, 2011. Spokane Valley-Rathdrum Prairie Aquifer Optimized Recharge for Summer Flow Augmentation of the Columbia River. Washington State Department of Ecology, Office of Columbia River, Yakima, Washington. http://www.ecy.wa.gov/geographic/spokane/ images/04012011_svrpa_orsfa.pdf

Bark, C.L., 2010. Assessing Diurnal and Seasonal Nitrate Behavior in Large Rivers using Nitratax In-Situ Sensors. MS thesis, University of Iowa.

Bates, B.C., Z.W. Kundzewicz, S. Wu and J.P. Palutikof (eds), 2008. Climate Change and Water. Technical Paper of the Intergovernmental Panel on Climate Change, IPCC

Secretariat, Geneva, 210 pp. https://drive.google.com/file/d/0B1gFp6Ioo3akcFFFeGRRVFNYM0E/view?pli=1

Berger, C. and S. Wells, 2005. Lake Whatcom Water Quality Model. Technical Report EWR-03-05, Maseeh College of Engineering and Computer Science, Department of Civil and Environmental Engineering, Portland State University, Portland, Oregon. URL: http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1151&context=cengin_fac

Berger, C., R.L. Annear and S.A. Wells, 2001. Lower Willamette River Model: Model Calibration. Technical Report EWR-2-01, Department of Civil Engineering, Portland State University, Portland, Oregon, 100 pages. URL: http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1145&context=cengin_fac

Berger, C., R.L. Annear and S.A. Wells, 2002. Upper Spokane River Model: Model Calibration, 1991 and 2000. Technical Report EWR-01-02, Department of Civil Engineering, Portland State University, Portland, Oregon. URL: http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1146&context=cengin_fac

Berger, C., R.L. Annear and S.A. Wells, 2003. Upper Spokane River Model: Model Calibration, 2001. Technical Report EWR-01-03, Department of Civil Engineering, Portland State University, Portland, Oregon. URL: http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1147&context=cengin_fac

Berger, C., R.L. Annear and S.A. Wells, 2004. Review of Spokane River Model for Washington Department of Ecology. Technical Report EWR-01-04, Portland State University Department of Civil Engineering. URL: http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1157&context=cengin_fac

Berger, C.J. and S.A. Wells, 2008. Modeling the Effects of Macrophytes on Hydrodynamics. *Journal of Environmental Engineering* 134(9): 778-788.

Berger, C.J., R.L. Annear, M.L. McKillip, V. Wells and S.A. Wells, 2009. Modeling the Spokane River-Lake Roosevelt System. 33rd IAHR Congress: *Water Engineering for a Sustainable Environment*.

Bottcher, A.B., B J. Whiteley, A.I. James and J.H. Hiscock, 2012. Watershed Assessment Model (WAM): Model Use, Calibration, and Validation. *American Society of Agricultural and Biological Engineers* 55(4): 1367-1383.

Bouraoui, F., L. Galbiati and G. Bidoglio, 2002. Climate Change Impacts on Nutrient Loads in the Yorkshire Ouse Catchment (UK). *Hydrology and Earth System Sciences Discussions, Copernicus Publications* 6(2): 197-209.

Bowen, J. and J. Hieronymus, 2003. A CE-QUAL-W2 Model of Neuse Estuary for Total Maximum Daily Load Development. *Journal of Water Resources Planning and Management* 129(4): 283-294.

Bowes, M., J. Smith, H. Jarvie and C. Neal. 2008, Modelling of Phosphorous Inputs to Rivers from Diffuse and Point Sources. *Science of the Total Environment* 395(2-3): 125-138.

Bowie, G.L., W.B. Mills, D.B. Porcella, C.L. Campbell, J.R. Pagenkopf, G.L. Rupp, K.M. Johnson, P.W.H. Chan, S.A. Gherini and C.E. Chamberlin, 1985. Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling (second Edition). EPA/600/3-85/040. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, GA. 455 pp. URL: http://www.ecs.umass.edu/eve/background/Algae/Bowie%201985%20rates%20WQ%20model.pdf

Bryson Project, Temperature and Dissolved Oxygen Study. Accessed From URL: https://www.duke-energy.com/pdfs/WQ_DOTemp_Bryson_Final.pdf

Buccola, N.L., A.J. Stonewall, A.B. Sullivan, Y. Kim and S.A. Rounds, 2013. Development of CE-QUAL-W2 Models for the Middle Fork Willamette and South Santiam Rivers, Oregon. U.S. Geological Survey Open-File Report 2013-1186, 55 p. http://pubs.usgs.gov/of/2013/1186/pdf/ofr2013-1186.pdf

Bunch, B.W., B.E. Johnson and M.S. Sarruff, 2003. Panama Lakes Water Quality Modeling Study. U.S. Army Corps of Engineers. Washington, DC. ERDC/ELTR-03-5. https://docs.micanaldepanama.com/plan-maestro/Study_Plan/Environmental_and_Social/Baseline_Studies/Water_quality_modeling_study/0006-01.pdf

California Water Plan, 2009. Integrated Water Management. Bulletin 160-09. Volume 2-Resource Management Strategies, Chapter 17: Pollution Prevention. URL: http://www.waterplan.water.ca.gov/docs/cwpu2009/0310final/v2c17_pollutprevent_cwp2009.pdf

Carmichael, J.J., K.M. Strzepek and B. Minarik, 1996. Impacts of Climate Change and Seasonal Variability on Economic Treatment Costs: A Case Study of the Nitra River Basin, Slovakia. *International Journal of Water Resources Development* 12(2): 209-228.

Carr, G.M. and J.P. Neary, 2008. Water Quality for Ecosystem and Human Health. Second Edition. United Nations Environment Programme Global Environment Monitoring System (GEMS)/Water Programme. ISBN: 92-95039-51-7. http:// www.unep.org/gemswater/Portals/24154/publications/pdfs/water_quality_human_health.pdf

CCSP, 2008. The Effects of Climate Change on Agriculture, Land Resources, Water Resources, and Biodiversity in the United States. A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research. P. Backlund, A. Janetos, D. Schimel, J. Hatfield, K. Boote, P. Fay, L. Hahn, C. Izaurralde, B.A. Kimball, T. Mader, J. Morgan, D. Ort, W. Polley, A. Thomson, D. Wolfe, M.G. Ryan, S.R. Archer, R. Birdsey, C. Dahm, L. Heath, J. Hicke, D. Hollinger, T. Huxman, G. Okin, R. Oren, J. Randerson, W. Schlesinger, D. Lettenmaier, D. Major, L. Poff, S. Running, L. Hansen, D. Inouye, B.P. Kelly, L. Meyerson, B. Peterson, R. Shaw. U.S. Department of Agriculture, Washington, DC., USA, 362 pp. URL: http://www.usda.gov/oce/climate_change/ SAP4_3/ CCSPFinalReport.pdf

Cerco, C.F. and T. Cole, 1993. Three-Dimensional Eutrophication Model of Chesapeake Bay. *Journal of Environmental Engineering* 119(6): 1006-1025.

Chai, T. and R.R. Draxler, 2014. Root Mean Square Error (RMSE) or Mean Absolute Error (MAE)? – Arguments against Avoiding RMSE in the Literature. *Geoscientific Model Development* 7: 1247–1250.

Chang, C., L. Cai, T. Lin, C. Chung, L. van der Linden and M. Burch, 2015. Assessment of the Impacts of Climate Change on the Water Quality of a Small Deep Reservoir in a Humid-Subtropical Climatic Region. *Water* 7(4): 1687-1711.

Chang, H., 2004. Water Quality Impacts of Climate and Land Use Changes in Southeastern Pennsylvania. *The Professional Geographer* 56(2): 240-257.

Chang, H., B.M. Evans and D.R. Easterling, 2001. The Effects of Climate Change on Stream Flow and Nutrient Loading. *Journal of the American Water Resources Association* 37(4): 973-985.

Chung, S. and R. Gu, 1998. Two-Dimensional Simulations of Contaminant Currents in Stratified Reservoir. *Journal of Hydraulic Engineering* 124(7): 704-711.

Colarusso, L.A., J.A. Chermak, J.C. Priscu and F.K. Miller, 2003. Modeling Pit Lake Water Column Stability using CE-QUAL-W2. *Proceedings of the Tailings and Mine Waste Conference*, 12-15 October 2003. Vail, Colorado. USA.

Cole, T.M. and D.H. Tillman, 1999. Water Quality Modeling of Lake Monroe using CE-QUAL-W2. U.S. Army Corps of Engineers Waterways Experiment Station. Paper MP-EL-99-1. http://el.erdc.usace.army.mil/elpubs/pdf/mpel99-1.pdf

Cole, T.M. and S.A. Wells, 2000. CE-QUAL-W2: A Two-Dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model, Version 3.0, User Manual. Instruction Report EL-00-1. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS. URL: http://www.cee.pdx.edu/w2/download/download/v30/ manual/ not%20online/parts1_3.pdf

Cole, T.M. and S.A. Wells, 2002. CE-QUAL-W2: A Two-Dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model, Version 3.1. Instruction Report EL-02-1, US Army Corps of Engineers, Washington, DC. 615 p

Cole, T.M. and S.A. Wells. 2003, CE-QUAL-W2: A Two-Dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model, Version 3.1. Instruction Report EL-03-1, US Army Engineering and Research Development Center, Vicksburg, MS. http:// pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1139&context=cengin_fac

Cox, B.A. and P.G. Whitehead, 2009. Impacts of Climate Change Scenarios on Dissolved Oxygen in the River Thames, UK. *Hydrology Research* 40(2-3): 138-152.

Crossman, J., M.N. Futter, S.K. Oni, P.G. Whitehead, L. Jin, D. Butterfield, H.M. Baulch and P.J. Dillon, 2013. Impacts of Climate Change on Hydrology and Water Quality: Future Proofing Management Strategies in the Lake Simcoe Watershed, Canada. *Journal of Great Lakes Research* 39(1): 19-32.

Cusimano, B., 2002. Spokane River Preliminary CE-QUAL-W2 Model Results. Environmental Assessment Program. Department of Ecology. http://www.ecy.wa.gov/programs/wq/tmdl/spokaneriver/dissolved_oxygen/spokane_do-rpt.pdf

Cusimano, B., 2003. Data Summary: Spokane River and Lake Spokane (Long Lake) Pollutant Loading Assessment for Protecting Dissolved Oxygen. Washington State Department of Ecology. Publication number 03-03-023. https://fortress.wa.gov/ecy/publications/documents/0303023.pdf

Cusimano, B., 2004. Spokane River and Lake Spokane (Long Lake) Pollutant Loading Assessment for Protecting Dissolved Oxygen. Washington State Department of Ecology. Publication No. 04-03-006. https://fortress.wa.gov/ecy/publications/documents/0403006.pdf

Donigian, A.S. Jr., 2002. Watershed Model Calibration and Validation: The HSPF Experience. *In Proceedings of WEF National TMDL Science and Policy* 44-73. Alexandria, Va.: Water Environment Federation.

Du, X., X. Li, W. Zhang and H. Wang, 2014. Variations in Source Apportionments of Nutrient Load among Seasons and Hydrological Years in a Semi-Arid Watershed: GWLF model results. *Environmental Science and Pollution Research* 21(10): 6506-6515.

Ebrahimi, M., E. Jabbari and H. Abbasi, 2015. Simulation of Thermal Stratification and Salinity in Dam Reservoir Using CE-QUAL-W2 Software (Case Study: Baft Dam). *Journal of Civil Engineering and Urbanism* 5(1): 07-11.

Ecology, 2012. Spokane River and Lake Spokane Dissolved Oxygen Total Maximum Daily Load. Water Quality Improvement Report.

Edinger, J.E., D.W. Duttweiler and J.C. Geyer, 1968. The Response of Water Temperatures to Meteorological Conditions. *Water Resources Research* 4(5): 1137-1143.

Edmondson, W.T., 1996. The Uses of Ecology: Lake Washington and Beyond. University of Washington Press, Seattle and London, p. 204. ISBN: 978-0-295-97569-6.

Erickson, T.R. and H.G. Stefan, 1996. Correlation of Oklahoma Stream Temperatures with Air Temperatures, Proj. Rep. 398, St. Anthony Falls Lab., Univ. of Minn., Minneapolis. https://conservancy.umn.edu/bitstream/handle/11299/109509/pr398.pdf?sequence=1

Fenelon, J.M., 1998. Water Quality in the White River Basin, Indiana, 1992-96. US Geological Survey. Circular 1150. URL: http://pubs.usgs.gov/circ/circ1150/pdf/circular1150.pdf

Ficke, A.D., C.A. Myrick and L.J. Hansen, 2007. Potential Impacts of Global Climate Change on Freshwater Fisheries. *Reviews in Fish Biology and Fisheries* 17(4): 581-613.

Field, C.B., L.D. Mortsch,, M. Brklacich, D.L. Forbes, P. Kovacs, J.A. Patz, S.W. Running and M.J. Scott, 2007. North America. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden and C.E. Hanson, (eds.). Cambridge University Press, Cambridge, UK, 617-652. p. 629.

Fisher, B.E.A. and G.W. Stewart, 2006. Impacts of Proposed Housing Growth in South and East England: Water Quality. *Science Summary* SC040047/SR5.

Flanagan, D.C., J.R. Frankenberger and J.C. Ascough II, 2012. WEPP: Model Use, Calibration, and Validation. *American Society of Agricultural and Biological Engineers* 55(4): 1463-1477.

Flowers, J.D., L.M. Hauck and R.L. Kiesling, 2001. Water Quality Modeling of Lake Waco Using CE-QUAL-W2 for Assessment of Phosphorus Control Strategies. Texas Institute for Applied Environmental Research. Tartleton State University. URL: http://tiaer.tarleton.edu/pdf/TR0114.pdf

Foster, J., A. Lowe and S. Winkelman, 2011. The Value of Green Infrastructure for Urban Climate Adaptation. The Center for Clean Air Policy. http://www.amwa.net/ galleries/climate-change/Green_Infrastructure_FINAL.pdf

Fu, G., 2005. Modeling Water Availability and its Response to Climatic Change for the Spokane River Watershed. Ph.D. thesis, Washington State University, Pullman, Wash.

Fu, G., M. Barber and S. Chen, 2007. Impacts of climate Change on Regional Hydrological Regimes in the Spokane River Watershed. *Journal of Hydrologic Engineering* 12(5): 452-461.

Fu, G., S.P. Charles and F.H.S. Chiew, 2007b. A Two-parameter Climate Elasticity of Streamflow Index to Assess Climate Change Effects on Annual Streamflow. *Water Resources Research* 43(11): 1-12. W11419.

Funk, W., H. Gibbons, R. Duffner, T. Notestine and T. Nielson, 1983. Water Quality of the Upper Spokane River and Evaluation of Methods for Measurement of the Effect of effluent upon Primary and Secondary Producers. Washington State Department of Ecology. Publication number 83-e11. URL: https://fortress.wa.gov/ecy/publications/publications/ 83e11.pdf

Galloway, J.M. and W.R. Green, 2002. Simulation of Hydrodynamics, Temperature, and Dissolved Oxygen in Norfork Lake, Arkansas, 1994-1995. U.S. Geological Survey Water-Resources Investigations Report 2002-4250. URL: http://pubs.usgs.gov/wri/2002/4250/report.pdf

Gearhart, C. and J.P. Buchanan, 2000. The Hydraulic Connection between the Spokane River and the Spokane Aquifer: Gaining and Losing Reaches of the Spokane River from Stateline, Idaho to Spokane. Washington. Spokane County Public Works Water Quality Management Program, 106 p.

Gelda, R. and S. Effler, 2007. Modeling Turbidity in a Water Supply Reservoir: Advancements and Issues. *Journal of Environmental Engineering* 133(2): 139-148.

Gelda, R., S. Effler and F. Peng, 2012. Modeling Turbidity and the Effects of Alum Application for a Water Supply Reservoir. *Journal of Environmental Engineering* 138(1): 38-47.

GeoEngineers, Inc., 2011. Deep Creek Field Data Collection Area, Bi-State Nonpoint Source Phosphorus Study Spokane County, Washington. File No. 0188-135-01. http://www.spokanecounty.org/data/utilitieswqmp/nps_documents/NPS%20Study_%20Deep%20Creek%20Data%20Collection_Final.pdf

Ghane, E., N.R. Fausey and L.C. Brown, 2015. Modeling Nitrate Removal in a Denitrification Bed. *Water Research* 71: 294-305.

Gibbons, H., W. Funk, R. Duffner, T. Nielson and T. Notestine, 1984. Baseline Study to Determine the Water Quality and the Primary and Secondary Producers of the Spokane River, Phase I. Washington State Department of Ecology. Publication number 84-e06. https://fortress.wa.gov/ecy/publications/documents/84e06.pdf

Gleick, P. H., (lead author) and others, 2000. Water: The Potential Consequences of Climate Variability and Change for the Water Resources of the United States. A report of the National Water Assessment Group for the U.S. Global Change Research Program. Pacific Institute for Studies in Development, Environment, and Security, Oakland, CA, USA. http://stephenschneider.stanford.edu/Publications/PDF_Papers/Gleick2000.pdf

Golosov, S., A. Terzhevik, I. Zverev, G. Kirillin and C. Engelhardt, 2012. Climate Change Impact on Thermal and Oxygen Regime of Shallow Lakes. *Tellus A* 64, 17264.

Grafton, R.Q., J. Pittock, R. Davis, J. Williams, G. Fu, M. Warburton, B. Udall, R. McKenzie, X. Yu, N. Che, D. Connell, Q. Jiang, T. Kompas, A. Lynch, R. Norris, H. Possingham and J. Quiggin, 2013. Global Insights into Water Resources, Climate Change and Governance. *Nature Climate Change* 3(4): 315-321.

Green, T.R., M. Taniguchi, H. Kooi, J.J. Gurdak, D.M. Allen, K.M. Hiscock, H. Treidel and A. Aureli, 2011. Beneath the Surface of Global Change: Impacts of Climate Change on Groundwater. *Journal of Hydrology* 405(3-4): 532-560.

Groisman, P.Y., R.W. Knight and T.R. Karl, 2001. Heavy Precipitation and High Streamflow in the Contiguous United States: Trends in the Twentieth Century. *Bulletin of the American Meteorological Society* 82(2): 219-246.

Ha, S.R. and J.Y. Lee, 2008. Application of CE-QUAL-W2 Model to Eutrophication Simulation in Daecheong Lake Stratified by Turbidity Storms. *In Proceedings of Taal 2007: The 12th World Lake Conference*, pp.824-833. M. Sengupta and R. Dalwani (eds.).

Haddad, O., P. Ashofteh, M. Ali-Hamzeh and M. Mariño, 2015. Investigation of Reservoir Qualitative Behavior Resulting from Sudden Entry of Biological Pollutant. *Journal of Irrigation and Drainage Engineering* 141(8), 04015003.

Hadjikakou, M., P.G. Whitehead, L. Jin, M. Futter, P. Hadjinicolaou and M. Shahgedanova, 2011. Modelling Nitrogen in the Yeşilirmak River Catchment in Northern Turkey: Impacts of Future Climate and Environmental Change and Implications for Nutrient Management. *Science of the Total Environment* 409(12): 2404-2418.

Hamlin-Tillman, D.E. and C. Haake, 1990. Predicted Water Quality Impacts of Impoundments on the Rogue River, Oregon. U.S. Army Engineer Waterways Experiment Station, Technical Report EL-90-5. http://soda.sou.edu/awdata/051007a1.pdf

Hammond, A.J., 2004. Hydrodynamic and Water Quality Simulation of Fecal Coliforms in the Lower Appomattox River, Virginia. Master’s Thesis, Virginia Polytechnic Institute and State University.

Hanna, R., 2014. Implementation of a Sediment Transport Model for CE-QUAL-W2. Portland State University. Civil and Environmental Engineering Master's Project Reports. Paper 8. http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1011&context=cengin_gradprojects

Hanna, R.B. and S.G. Campbell, 2000. Water Quality Modeling in the Systems Impact Assessment Model for the Klamath River Basin- Keno, Oregon to Seiad Valley, California, Open-File Report 99-113 U.S. Department Of The Interior, U.S. Geological Survey. URL: https://www.fort.usgs.gov/sites/default/files/products/publications/21130/ 21130.pdf

Hart, R.M., W.R. Green, D.A. Westerman, J.C. Petersen and J.L. De Lanois, 2012. Simulated Effects of Hydrologic, Water Quality, and Land-Use Changes of the Lake Maumelle Watershed, Arkansas, 2004–10. U.S. Geological Survey Scientific Investigations Report 2012–5246, 119 p. (Revised February 2013). URL: http:// pubs.usgs.gov/sir/2012/5246/

Hashim, W. and H. Bresler, 2005. Washington's Water Quality Management Plan to Control Nonpoint Source Pollution, 2005. Washington State Department of Ecology. Publication number 05-10-027. https://fortress.wa.gov/ecy/publications/publications/

0510027.pdf

Hatano, R., T. Nagumo, H. Hata and K. Kuramochi, 2005. Impact of Nitrogen Cycling on Stream Water Quality in a Basin Associated with forest, Grassland, and Animal Husbandry, Hokkaido, Japan. *Ecological Engineering* 24(5): 509-515.

Hatfield, J., K. Boote, P. Fay, L. Hahn, C. Izaurralde, B.A. Kimball, T. Mader, J. Morgan, D. Ort, W. Polley, A. Thomson and D. Wolfe, 2008. Agriculture. In: The Effects of Climate Change on Agriculture, Land Resources, Water Resources, and Biodiversity in the United States. A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research. Washington, DC., USA, 362 pp. http://www.usda.gov/oce/climate_change/SAP4_3/CCSPFinalReport.pdf

Hodgkins, G.A., R.W. Dudley and T.G. Huntington, 2005. Summer Low Flows in New England during the 20th Century. *Journal of the American Water Resources Association* 41(2): 403-411.

Hortness, J.E. and J.J. Covert, 2005. Streamflow Trends in the Spokane River and tributaries, Spokane Valley/Rathdrum Prairie, Idaho and Washington. U.S. Geological Survey Scientific Investigations Report 2005-5005, 17 p. http://pubs.usgs.gov/sir/2005/5005/pdf/sir20055005.pdf

Hostetler, S.W., 1991. Analysis and Modeling of Long-term Stream Temperature on the Steamboat Creek Basin, Oregon: Implications for Land Use and Fish Habitat. *Water Resources Bulletin* 27(4): 637-647.

Idaho Department of Environmental Quality, 1999. Coeur D'Alene Lake and River Sub-basin Assessment and Proposed Maximum Daily Loads. https://www.deq.idaho.gov/media/453634-_water_data_reports_surface_water_tmdls_cda_lake_river_cda_lake_river_entire.pdf

Idaho Department of Environmental Quality, 2013. Coeur D'Alene Lake and River Subbasin Assessment and Total Maximum Daily Loads. http://www.deq.idaho.gov/media/1040175-fernan_lake_addendum_draft_0913.pdf

IPCC, 2007. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. URL: https://www.ipcc.ch/pdf/assessment-report/ar4/wg1/ar4_wg1_full_report.pdf

IPCC, 2012. Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change. Field, C.B., V. Barros, T.F. Stocker, D. Qin, D.J. Dokken, K.L. Ebi, M.D. Mastrandrea, K.J. Mach, G.-K. Plattner, S.K. Allen, M. Tignor and P.M.

Midgley (eds.). Cambridge University Press, Cambridge, UK, and New York, NY, USA, 582 pp. https://www.ipcc-wg2.gov/SREX/images/uploads/SREX-All_FINAL.pdf

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. T.F. Stocker, D. Qin, G.K. Plattner, M. Tignor, K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. 1535 pp. URL: https://www.ipcc.ch/pdf/assessment-report/ar5/wg1/WGIAR5_SPM_brochure_en.pdf

Jaber, F.H. and S. Shukla, 2012. MIKE-SHE: Model Use, Calibration, and Validation. *American Society of Agricultural and Biological Engineers* 55(4): 1479-1489.

Jakeman, A.J., R.A. Letcher and J.P. Norton, 2006. Ten iterative steps in development and evaluation of environmental models. *Environmental Modelling & Software* 21(5): 602-614.

Jeznach, L.C. and J.E. Tobiason, 2015. Future Climate Effects on Thermal Stratification in the Wachusett Reservoir. *Journal - American Water Works Association* 107(4): 197-209.

Jin, X. and V. Sridhar, 2012. Impacts of Climate Change on Hydrology and Water Resources in the Boise and Spokane River Basins. *Journal of the American Water Resources Association* 48(2): 197-220.

Johnson, F.A., 1971. Stream Temperatures in an Alpine Area. *Journal of Hydrology* 14(3-4): 322-336.

Karl, T.R., J.M. Melillo and T.C. Peterson (eds.), 2009. Global Climate Change Impacts in the United States. United States Global Change Research Program (USGCRP). Cambridge University Press, New York, NY, USA. https://downloads.globalchange.gov/ usimpacts/ pdfs/climate-impacts-report.pdf

Kaste, Ø., R.F. Wright, L.J. Barkved, B. Bjerkeng, T. Engen-Skaugen, J. Magnusson and N.R. Sælthun, 2006. Linked Models to Assess the Impacts of Climate Change on Nitrogen in a Norwegian River Basin and Fjord System. *Science of the Total Environmen*t 365(1-3): 200-222.

Katrina, P., H. Greig, P. Thompson, T. Carvalho-Pereira and J. Shurin, 2012. Warming Modifies Trophic Cascades and Eutrophication in Experimental Freshwater Communities. *Ecology* 93: 1421-1430.

Khangaonkar, T., B. Sackmann, W. Long, T. Mohamedali and M. Roberts, 2012. Puget Sound Dissolved Oxygen Modeling Study: Development of an Intermediate Scale Water Quality Model. Prepared for the U.S. Department of Energy. PNNL-20384. http:// www.ecy.wa.gov/programs/wq/PugetSound/PSWQModelRpt082011TK02.pdf

Klose, K., S. Cooper, A. Leydecker and J. Kreitler, 2012. Relationships among Catchment Land Use and Concentrations of Nutrients, Algae, and Dissolved Oxygen in a Southern California River. *Freshwater Science* 31(3): 908-927.

Knight, D.T., 1998. Watershed Approach to Water Quality Management Needs Assessment for the Spokane Water Quality Management Area (WRIA 54-57). Washington State Department of Ecology, Water Quality Program, Spokane, WA.

Kundzewicz, Z.W., L.J. Mata, N.W. Arnell, P. Döll, P. Kabat, B. Jiménez, K.A. Miller, T. Oki, Z. Sen and I.A. Shiklomanov, 2007. Freshwater Resources and Their Management. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden and C.E. Hanson, eds., Cambridge University Press, Cambridge, UK, 173-210. URL: https://www.ipcc.ch/pdf/assessment-report/ar4/wg2/ar4-wg2-chapter3.pdf

Kuo, J., W. Lung, C. Yang, W. Liu, M. Yang and T. Tang, 2006. Eutrophication Modelling of Reservoirs in Taiwan. *Environmental Modelling & Software* 21(6): 829-844.

Lee, K.E., D.L. Lorenz, J.C. Petersen and J.B. Greene, 2012. Seasonal Patterns in Nutrients, Carbon, and Algal Responses in Wadeable Streams within Three Geographically Distinct Areas of the United States, 2007–08. U.S. Geological Survey Scientific Investigations Report 2012–5086, 55 p. URL: http://pubs.usgs.gov/sir/2012/5086/sir12-5086.pdf

Lettenmaier, D., D. Major, L. Poff and S. Running, 2008. Water Resources. In: The Effects of Climate Change on Agriculture, Land Resources, Water Resources, and Biodiversity in the United States. A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research. Washington, DC., USA, 362 pp. http://www.usda.gov/oce/climate_change/SAP4_3/CCSPFinalReport.pdf

Lillebø, A.M., M. Morais, P. Guilherme, R. Fonseca, a. Serafim and R. Neves, 2007. Nutrient Dynamics in Mediterranean Temporary Streams: A Case Study in Pardiela Catchment (Degebe River, Portugal). *Limnologica - Ecology and Management of Inland Waters* 37(4): 337–348.

Lin, J., X. L. Xie, L.J. Pietrafesa, J. Shen, M.A. Mallin and M.J. Durako, 2006. Dissolved Oxygen Stratification in Two Micro-tidal Partially-mixed Estuaries. *Estuarine, Coastal and Shelf Science* 70(3): 423-437.

Liu, J.P., M.R. Song, R.M. Horton and Y.Y. Hu, 2013. Reducing Spread in Climate Model Projections of a September Ice-free Arctic *Proceedings of the National Academy of Sciences* USA 110 12571–6

Ludwig, F., P. Kabat, H. van Schaik and M. van der Valk, 2009. Climate Change Adaptation in the Water Sector. Earthscan, London. ISBN: 978-1-84407-652-9.

Ma, J., D. Liu, S.A. Wells, H. Tang, D. Ji and Z. Yang, 2015. Modeling Density Currents in a Typical Tributary of the Three Gorges Reservoir, China. *Ecological Modelling* 296: 113-125.

Macintosh, K.A., P. Jordan, R. Cassidy, J. Arnscheidt and C. Ward, 2011. Low Flow Water Quality in Rivers; Septic Tank Systems and High-Resolution Phosphorus Signals. *Science of the Total Environment* 412-413: 58-65.

Maloney, E.D., S.J. Camargo, E. Chang, B. Colle, R. Fu, K.L. Geil, Q. Hu, X. Jiang, N. Johnson, K.B. Karnauskas, J. Kinter, B. Kirtman, S. Kumar, B. Langenbrunner, K. Lombardo, L.N. Long, A. Mariotti, J.E. Meyerson, K.C. Mo, J.D. Neelin, Z. Pan, R. Seager, Y. Serra, A. Seth, J. Sheffield, J. Stroeve, J. Thibeault, S. Xie, C. Wang, B. Wyman and M. Zhao, 2014. North American Climate in CMIP5 Experiments: Part III: Assessment of Twenty-First-Century Projections. *Journal of Climate* 27(6): 2230-2270.

Martin, J., 1988. Application of Two-Dimensional Water Quality Model. *Journal of Environmental Engineering* 114(2): 317-336.

Maryland DNR. Chesapeake Bay Monitoring: Monitoring for Management Actions. http://www.dnr.state.md.us/bay/monitoring/mon_mngmt_actions/chapter7.html

McCulloch, A.J., 2011. Developing and Calibrating the Hydrodynamic and Water Quality Model CE-QUAL-W2 for Banks Lake Washington. Master's Thesis. Portland State University.

McIntyre, N.R., T. Wagener, H.S. Wheater and S.C. Chapra, 2003. Risk-based Modelling of Surface Water Quality: A Case Study of the Charles River, Massachusetts. *Journal of Hydrology* 274(1-4): 225-247.

Miao, C., Q. Duan, Q. Sun, Y. Huang, D. Kong, T. Yang, A. Ye, Z. Di and W. Gong, 2014. Assessment of CMIP5 Climate Models and Projected Temperature Changes Over Northern Eurasia. *Environmental Research Letters* 9(5): 055007.

Millican, J., J. Back and A. McFarland, 2008. Nutrient Bioassays of Growth Parameters for Algae in the North Bosque River of Central Texas. *Journal of the American Water Resources Association* 44(5): 1219-1230.

Miserendino, M.L., R. Casaux, M. Archangelsky, C.Y. Di Prinzio, C. Brand and A.M. Kutschker, 2011. Assessing Land-use Effects on Water Quality, In-Stream Habitat, Riparian Ecosystems and Biodiversity in Patagonian Northwest Streams. *Science of the Total Environment* 409(3): 612-624.

Mitchell, T.D., T.R. Carter, P.D. Jones, M. Hulme and M. New, 2004. A Comprehensive Set of High-Resolution Grids of Monthly Climate for Europe and the Globe: The Observed Record (1901-2000) and 16 Scenarios (2001-2100). Tyndall Centre Working Paper, 55. 37 pp. http://ipcc-data.org/docs/tyndall_working_papers_wp55.pdf

Modiri-Gharehveran, M., A.E. Shahidi and E. Jabbari, 2014. Effects of Climate Change on the Thermal Regime of a Reservoir. *Water Management* 167(10): 601-611.

Mohseni, O., H.G. Stefan and T.R. Erickson, 1998. A Nonlinear Regression Model for Weekly Stream Temperatures. *Water Resources Research* 34(10): 2685–2692.

Mohseni, O., T.R. Erickson and H.G. Stefan, 1999. Sensitivity of Stream Temperature in the United States to Air Temperature Projected under a Global Warming Scenario. *Water Resources Research* 35(12): 3723-3733.

Moore, D. and J. Ross, 2007. Spokane River and Lake Spokane Dissolved Oxygen Total Maximum Daily Load, Water Quality Improvement Report. Washington State Department of Ecology. Publication number 07-10-073.

Moore, D.J. and J. Ross, 2010. Spokane River and Lake Spokane Dissolved Oxygen Total Maximum Daily Load, Water Quality Improvement Report. Washington State Department of Ecology Eastern Regional Office, Spokane, Washington. Publication No. 07-10-073. https://fortress.wa.gov/ecy/publications/documents/0710073.pdf

Moriasi D.N., B.N. Wilson, K.R. Douglas-Mankin, J.G. Arnold and P.H. Gowda, 2012. Hydrologic and Water Quality Models: Use, Calibration, and Validation. *American Society of Agricultural and Biological Engineers* 55(4): 1241-1247.

Moriasi, D.N., J.G. Arnold, M.W. Van Liew, R.L. Bingner, R.D. Harmel and T.L. Veith, 2007. Model Evaluation Guidelines for Systematic Quantification of Accuracy in Watershed Simulations. *Transactions ASABE* 50(3): 885-900.

Mote P., E. Salathé, V. Dulière and E. Jump, 2008. Scenarios of Future Climate for the Pacific Northwest. Climate Impacts Group, University of Washington. URL: http://cses.washington.edu/db/pdf/moteetal2008scenarios628.pdf

Mote, P., L.Brekke, P. B.Duffy and E.Maurer, 2011. Guidelines for Constructing Climate Scenarios, *Eos, Transactions American Geophysical Union* 92(31): 257-264.

Mote, P.W. and E.P. Salathé, 2009. Future Climate in the Pacific Northwest. Chapter 1: In Washington Climate Change Impacts Assessment: Evaluating Washington's Future in a Changing Climate, Climate Impacts Group, University of Washington, Seattle, Washington. http://www.cses.washington.edu/db/pdf/wacciaexecsummary638.pdf

Mote, P.W. and E.P. Salathé, 2010. Future Climate in the Pacific Northwest. *Climatic Change* 102(1-2): 29-50.

Mote, P.W., J.T. Abatzoglou and K.E. Kunkel, 2013. Climate: Variability and Change in the Past and the Future. Chapter 2, 25-40, in M.M. Dalton, P.W. Mote, and A.K. Snover (eds.) Climate Change in the Northwest: Implications for Our Landscapes, Waters, and

Communities. Washington D.C.: Island Press. http://cses.washington.edu/db/pdf/daltonetal678.pdf

Nash, J.E. and J.V. Sutcliffe, 1970. River Flow Forecasting through Conceptual Models, I-A, Discussion of Principles. *Journal of Hydrology* 10(3): 282-290.

Neff, R., H. Chang, C.G. Knight, R.G. Najjar, B. Yarnal and H.A. Walker, 2000. Impact of Climate Variation and Change on Mid-Atlantic Region Hydrology and Water Resources. *Climate Research* 14: 207-218.

New M., M. Hulme and P. Jones, 2000. Representing Twentieth-Century Space–Time Climate Variability. Part II: Development of 1901–96 Monthly Grids of Terrestrial Surface Climate. *Journal of Climate* 13(13): 2217–2238.

Nielsen, E.J., 2005. Algal Succession and Nutrient Dynamics in Elephant Butte Reservoir. Brigham Young University, All Theses and Dissertations. Paper 280.

Nõges, T., L. Tuvikene and P. Nõges, 2010. Contemporary trends of Temperature, Nutrient Loading and Water Quality in Large Lakes Peipsi and Võrtsjärv, Estonia. *Aquatic Ecosystem Health & Management* 13(2): 143-153.

Northwest Power and Conservation Council, 2004. Spokane Subbasin Plan. http://www.nwcouncil.org/fw/subbasinplanning/admin/level2/intermtn/plan/spokane.pdf

Novotny, E.V. and H.G. Stefan, 2007. Stream Flow in Minnesota: Indicator of Climate Change. *Journal of Hydrology* 334(3): 319-333.

Paerl, H.W. and V.J. Paul, 2012. Climate Change: Links to Global Expansion of Harmful Cyanobacteria. *Water Research* 46(5): 1349-1363.

Pastuszak, M., 1995. The Hydrochemical and Biological Impact of the River Vistula on the Pelagic System of the Gulf of Gdańsk in 1994. Part 1. Variability in nutrient concentrations. *Oceanologia* 37(2): 181-205.

Patmont, C.R., G.J. Pelletier and M.E. Harper, 1985. Phosphorus Attenuation in the Spokane River. Project Completion Report. Contract No. C84-076. Harper-Owes, Seattle, WA Prepared for Washington State Department of Ecology, Olympia, WA. Publication No. 85-e28.

Patmont, C.R., G.J. Pelletier, L. Singleton, R. Soltero, W. Trial and E. Welch, 1987. The Spokane River Basin: Allowable Phosphorus Loading. Final Report. Contract No. C0087074. Harper-Owes, Seattle, WA. Prepared for State of Washington State Department of Ecology, Olympia, WA. Publication No. 87-e29.

Pelletier, G., 1994. Dissolved Oxygen in the Spokane River Downstream from Inland Empire Paper Company with Recommendations for Waste Load Allocations for

Biochemical Oxygen Demand. Washington State Department of Ecology. Publication number 94-155. https://fortress.wa.gov/ecy/publications/documents/94155.pdf

Pelletier, G., 1997. Waste Load Allocations for Biochemical Oxygen Demand for Inland Empire Paper Company. Washington State Department of Ecology. Publication number 97-313. https://fortress.wa.gov/ecy/publications/documents/97313.pdf

Perry, R.W., J.C. Risley, S.J. Brewer, E.C. Jones and D.W. Rondorf, 2011. Simulating Daily Water Temperatures of the Klamath River under Dam Removal and Climate Change Scenarios. U.S. Geological Survey Open-File Report 2011-1243, 78 p. http://pubs.usgs.gov/of/2011/1243/pdf/ofr20111243.pdf

Pilgrim, J.M. and H.G. Stefan, 1995. Correlation of Minnesota stream Water Temperatures with Air Temperatures, Proj. Rep. 382, St. Anthony Falls Lab., Univ. of Minn., Minneapolis. Accessed from URL: https://conservancy.umn.edu/bitstream/handle/11299/109299/pr382.pdf?sequence=1

Pionke, H.B., W.J. Gburek, R.R. Schnabel, A.N. Sharpley and G.F. Elwinger, 1999. Seasonal Flow, Nutrient Concentrations and Loading Patterns in Stream Flow Draining an Agricultural Hill-land Watershed. *Journal of Hydrology* 220(1–2): 62-73.

Rau, B., 2015. Washington's Water Quality Management Plan to Control Nonpoint Sources of Pollution. Washington State Department of Ecology. Publication number 15-10-015. https://fortress.wa.gov/ecy/publications/documents/1510015.pdf

Refsgaard, J.C., 1997. Parameterization, Calibration, and Validation of Distributed Hydrological Models. *Journal of Hydrology* 198(1-4): 69-97.

Rounds, S.A. and N.L. Buccola, 2015. Improved Algorithms in the CE-QUAL-W2 Water-quality Model for Blending Dam Releases to meet Downstream Water-temperature Targets. U.S. Geological Survey Open-File Report 2015-1027, 40 p. http://pubs.usgs.gov/of/2015/ 1027/pdf/ofr2015-1027.pdf

Rowe, L.K. and C.H. Taylor, 1994. Hydrology and Related Changes after Harvesting Native Forest Catchments and Establishing *Pinus radiata* Plantations, Part 3, Stream Temperatures. *Hydrological Processes* 8(4): 299-310.

Rupp D.E., J.T. Abatzoglou, K.C. Hegewisch and P.W. Mote, 2013. Evaluation of CMIP5 20th Century Climate Simulations for the Pacific Northwest USA. *Journal of Geophysical Research: Atmospheres* 118(19): 10884–10906.

Samal, N. R., A.H. Matonse, R. Mukundan, M.S. Zion, D.C. Pierson, R.K. Gelda and E.M. Schneiderman, 2013. Modelling potential Effects of Climate Change on Winter Turbidity Loading in the Ashokan Reservoir, NY. *Hydrological Processes* 27(21): 3061–3074.

Santhi, C., J.G. Arnold, J.R. Williams, W.A. Dugas, R. Srinivasan and L.M. Hauck, 2001.

Validation of the SWAT Model on a Large River Basin with Point and Nonpoint Sources. *Journal of the American Water Resources Association* 37(5): 1169-1188.

Sarmiento, J.L. and N. Gruber, 2005. Ocean Biogeochemical Dynamics. Princeton University Press. Special Version Adapted For AOS105/EEB139. URL: http:// people.atmos.ucla.edu/gruber/teaching/book_aos105_w05/book-merged-ucla_05.pdf

Seitzinger, S., J.A. Harrison, J.K. Bohlke, A.F. Bouwman, R. Lowrance, B. Peterson, C. Tobias and G.V. Drecht, 2006. Denitrification across Landscapes and Waterscapes: A Synthesis. *Ecological Applications* 16(6): 2064-2090.

Sincock, A.M., H.S. Wheater and P.G. Whitehead, 2003. Calibration and Sensitivity Analysis of a River Water Quality Model under Unsteady Flow Conditions. *Journal of Hydrology* 277(3–4): 214-229.

Singh, J., H.V. Knapp and M. Demissie, 2004. Hydrologic Modeling of the Iroquois River Watershed using HSPF and SWAT. Illinois State Water Survey CR 2004-08. http:// swat.tamu.edu/media/90101/singh.pdf

Singleton, L., 1981. Spokane River Wasteload Allocation Study - Supplemental Report for Phosphorus Allocation. Washington State Department of Ecology. Publication number 81-15. https://fortress.wa.gov/ecy/publications/documents/8115.pdf

Singleton, V., B. Jacob, M. Feeney and J. Little, 2013. Modeling a Proposed Quarry Reservoir for Raw Water Storage in Atlanta, Georgia. *J. Environ. Eng.* 139(1): 70-78.

Slominski, S., R. Annear, C. Berger and S. Wells, 2002. Upper Spokane River Model: Boundary Conditions and Model Setup, 2001. Technical Report EWR-04-02, Department of Civil Engineering, Portland State University, Portland, Oregon. URL: http:// pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1150&context=cengin_fac

Smith, D.L., T.L. Threadgill and C.E. Larson, 2012. Modeling the Hydrodynamics and Water Quality of the Lower Minnesota River Using CE-QUAL-W2. U.S. Army Corps of Engineers. ERDC/EL TR-12-12. http://el.erdc.usace.army.mil/elpubs/pdf/trel12-12.pdf

Smith, E.A., R.L. Kiesling, J.M. Galloway and J.R. Ziegeweid, 2014. Water Quality and Algal Community Dynamics of Three Deepwater Lakes in Minnesota utilizing CE-QUAL-W2 Models. U.S. Geological Survey Scientific Investigations Report 2014–5066, 73 p. http:// pubs.usgs.gov/sir/2014/5066/pdf/sir2014-5066.pdf

Solheim, A.L., K. Austnes, T.E. Eriksen, I. Seifert and S. Holen, 2010. Climate Change Impacts on Water Quality and Biodiversity. European Environmental Agency. European Topic Centre, Water Technical Report. URL: http://icm.eionet.europa.eu/docs/ Climate_impacts_on_water_quality_and_biodiversity_29_Nov_2010_final_2.pdf

Spokane County. Draft Environmental Impact Statement, Spokane County Wastewater

Facilities Plan. http://www.spokanecounty.org/utilities/wwfp/Nov01Rpt/DraftEIS.htm

Staehr, P.A. and K. Sand-Jensen, 2006. Seasonal Changes in Temperature and Nutrient Control of Photosynthesis, Respiration and Growth of Natural Phytoplankton Communities. *Freshwater Biology* 51(2): 249-262.

Stakhiv, E. and B. Stewart. 2010. Needs for climate information in support of decision-making in the water sector. *Procedia Environmental Sciences* 1: 102-119.

Stefan, H.G. and B.A. Sinokrot, 1993. Projected Global Climate Change Impact on Water Temperatures in Five North Central U.S. Streams. *Climatic Change* 24(4): 353-381.

Stefan, H.G. and E.B. Preud'homme, 1993. Stream Temperature Estimation from Air Temperature. *Water Resources Research* 29(1): 27-45.

Stefan, H.G., J. Gulliver, M.G. Hahn and A.Y. Fu, 1980. Water Temperature Dynamics in Experimental Field Channels: Analysis and Modeling, Proj. Rep. 193, St. Anthony Falls Lab., Univ. of Minn., Minneapolis.

Stocker, T.F., D. Qin, G.K. Plattner, L.V. Alexander, S.K. Allen, N.L. Bindoff, F.M. Bréon, J.A. Church, U. Cubasch, S. Emori, P. Forster, P. Friedlingstein, N. Gillett, J.M. Gregory, D.L. Hartmann, E. Jansen, B. Kirtman, R. Knutti, K.K. Kumar, P. Lemke, J. Marotzke, V. MassonDelmotte, G.A. Meehl, I.I. Mokhov, S. Piao, V. Ramaswamy, D. Randall, M. Rhein, M. Rojas, C. Sabine, D. Shindell, L.D. Talley, D.G. Vaughan and S.P. Xie, 2013. Technical Summary In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA,

Stuart, M.E., D.C. Gooddy, J.P. Bloomfield and A.T. Williams, 2011. A Review of the Impact of Climate Change on Future Nitrate Concentrations in Groundwater of the UK. *Science of the Total Environment* 409(15): 2859-2873.

Stumm, W. and J.J. Morgan, 1981. Aquatic Chemistry: An Introduction Emphasizing Chemical Equilibria in Natural Waters. Wiley Interscience, New York, NY. 795 p. ISBN: 978-0-471-51185-4

Sullivan, A. and S. Rounds, 2005. Modeling Hydrodynamics, Temperature, and Water Quality in Henry Hagg Lake, Oregon, 2000–03, Prepared in cooperation with Clean Water Services, Scientific Investigations Report 2004–5261, Department of the Interior, Geological Survey. http://pubs.usgs.gov/sir/2004/5261/pdf/sir2004-5261.pdf

Sullivan, A.B., I.E. Sogutlugil, S.A. Rounds and M.L. Deas, 2013. Modeling the Water-quality Effects of Changes to the Klamath River Upstream of Keno Dam, Oregon. U.S. Geological Survey Scientific Investigations Report 2013–5135, 60 p. URL: http://pubs.usgs.gov/sir/2013/5135/pdf/sir20135135.pdf

Supharatid, S., 2015. Assessment of CMIP3-CMIP5 Climate Models Precipitation Projection and Implication of Flood Vulnerability of Bangkok. *American Journal of Climate Change* 4(1): 140-162.

Talley L., G. Pickard, W. Emery and J. Swift, 2011. Descriptive Physical Oceanography: An Introduction, Sixth Edition. Elsevier, Boston, 560 pp. ISBN: 978-0-7506-4552-2.

Taylor, K.E., R.J. Stouffer and G.A Meehl, 2012. An Overview of CMIP5 and the Experimental Design. *Bulletin of the American Meteorological Society* 93(4): 485-498.

Tillman, D.H., 1992. Predicted Water Quality Impacts from Reducing Flow out of Gavins Point Dam on the Missouri River. Army Engineer Waterways Experiment Stn., Vicksburg, MS, USA. WES/TR/EL-92-36. 184 pp.

Tong, S.T.Y., A.J. Liu and J.A. Goodrich, 2007. Climate Change Impacts on Nutrient and Sediment Loads in a Midwestern Agricultural Watershed. *Journal of Environmental Informatics* 9(1): 18-28.

Tsihrintzis, V.A., H.R. Fuentes and L. Rodriguez, 1995. Modeling Water Quality in Low Velocity Sub-tropical Streams. In Water Pollution III: Modelling, Measuring and Prediction. L.C. Wrobel and P. Latinopoulos (eds). Computational Mechanics Publications. pp. 171-178.

Tsvetkova, O.V., 2013. Uncertainty in Climatic Change Impacts on Multiscale Watershed Systems. Ph.D. Dissertations. University of Massachusetts Amherst. Paper 845.

Tu, J., 2008. Assessing the Impact of Long-term Land Use Changes on Water Quality in Eastern Massachusetts. Ph.D. Thesis, City University of New York, 271 pages; UMI 3325454.

Tu, J., 2009. Combined Impact of Climate and Land Use Changes on Streamflow and Water Quality in Eastern Massachusetts, USA. *Journal of Hydrology* 379(3-4): 268-283.

Tu, J., 2011. Spatially Varying Relationships between Land Use and Water Quality across an Urbanization Gradient Explored by Geographically Weighted Regression. *Applied Geography* 31(1): 376-392.

USEPA, U.S. Environmental Protection Agency, 2008. National Water Program Strategy, Response to Climate Change. EPA 800-R-08-001. Accessed from URL: http:// www.allianceforwaterefficiency.org/uploadedFiles/Resource_Center/Library/water_resources/National-Water-Program-Strategy-Response-to-Climate%20Change.pdf

USEPA, U.S. Environmental Protection Agency, 2010. Green Infrastructure Case Studies: Municipal Policies for Managing Stormwater with Green Infrastructure. EPA Office of Wetlands, Oceans and Watersheds. EPA-841-F-10-004.

van Orden, G.N. and C.G. Uchrin, 1993. The Study of Dissolved Oxygen Dynamics in the Whippany River, New Jersey using the QUAL2E Model. *Ecological Modelling* 70(1-2): 1-17.

van Vuuren, D.P., J. Edmonds, M. Kainuma, K. Riahi, A. Thomson, K. Hibbard, G.C. HurO, T. Kram, V. Krey, J.F. Lamarque, T. Masui, M. Heinshausen, N. Nakićenović, S.J. Smith and S.K. Rose, 2011. The Representative Concentration Pathways: An Overview. *Climate Change* 109(1): 5-31.

Verweij, W., J. van der Wiele, I. van Moorselaar and E. van der Grinten, 2010. Impact of Climate Change on Water Quality in the Netherlands. National Institute for Public Health and the Environment. Report 607800007. URL: http://www.rivm.nl/dsresource?objectid=rivmp:16510&type=org&disposition=inline&ns_nc=1

Wells, S. A., V.I. Wells and C.J. Berger, 2008. Water Quality and Hydrodynamic Modeling of Tenkiller Reservoir. Report prepared for the State of Oklahoma. Case No. 05-CU-329-GKF-SAJ. URL: https://www.owrb.ok.gov/quality/standards/pdf_standards/scenicrivers/OK%20Attorney% 20General%20-%20Wells%20File.pdf

Wells, S.A., 1999. River Basin Modeling Using CE-QUAL-W2 Version 3. Proc. *ASCE International Water Resources Engineering Conference* Seattle, WA, 1999.

Wells, S.A., R. Annear, C. Berger and M. Sytsma, 2000. Modeling and Analysis of Cooper Creek Reservoir Water Quality. City of Sutherlin, OR. 75 pages.

Wells, S.A., R.L. Annear and C.J. Berger, 2003. Upper Spokane River Model in Idaho: Boundary Conditions and Model Setup for 2001. Technical Report EWR-02-03. Department of Civil and Environmental Engineering, Portland State University. http://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1158&context=cengin_fac

Whitehead, P.G., R.L. Wilby, R.W. Battarbee, M. Kernan and A.J. Wade, 2009. A Review of the Potential Impacts of Climate Change on Surface Water Quality. *Hydrological Sciences Journal* 54(1): 101-123.

Willett, K.M., 2007. Creation and analysis of HadCRUH: A New Global Surface Humidity Dataset. Ph.D. thesis, University of East Anglia, Norwich, UK.

Williams, N.T., 2007. Modeling Dissolved Oxygen in Lake Powell using CE-QUAL-W2. Brigham Young University. All Theses and Dissertations. Paper 849.

Witte, J.P.M., J. Runhaar, R. van Ek, D.C.J. van der Hoek, R.P. Bartholomeus, O. Batelaan, P.M. van Bodegom, M.J. Wassen and S.E.A.T.M. van der Zee, 2012. An Ecohydrological Sketch of Climate Change Impacts on Water and Natural Ecosystems for the Netherlands: Bridging the Gap between Science and Society. *Hydrology and Earth System Sciences* 16(5): 3945-3957.

Woods, P.F., 1989. Hypolimnetic Concentrations of Dissolved Oxygen, Nutrients, and Trace Elements in Coeur D'alene Lake, Idaho. U.S. Geological Survey Water-Resources Investigations Report 89-4032. http://pubs.usgs.gov/wri/1989/4032/report.pdf

WSOFM, State of Washington Office of Financial Management, 2012. County Growth Management Population Projections by Age and Sex: 2010-2040. Forecasting Division. http://www.ofm.wa.gov/pop/gma/projections12/GMA_2012_county_pop_projections.pdf

Wu, J., R. Zou and S.L. Yu, 2006. Uncertainty Analysis for Coupled Watershed and Water Quality Modeling Systems. *Journal of Water Resources Planning and Management* 132(5): 351-361.

Yaeger, M.A., M. Housh, X. Cai and M. Sivapalan, 2014. An Integrated Modeling Framework for Exploring Flow Regime and Water Quality Changes with Increasing Biofuel Crop Production in the U.S. Corn Belt. *Water Resources Research* 50(12): 1944-7973.

Yu, D., P. Shi, Y. Liu and B. Xun, 2013. Detecting Land Use-water Quality Relationships from the Viewpoint of Ecological Restoration in an Urban Area. *Ecological Engineering* 53: 205-216.

Zhang, H, D.A. Culver and L. Boegman, 2008. A Two-Dimensional Ecological Model of Lake Erie: Application to Estimate Dreissenid Impacts on Large Lake Plankton Populations. *Ecological Modelling* 214(2-4): 219-241.

Zhang, Z., B. Sun and B.E. Johnson, 2015. Integration of a Benthic Sediment Diagenesis Module into the Two Dimensional Hydrodynamic and Water Quality Model - CE-QUAL-W2. *Ecological Modelling* 297: 213-231.